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PERFORMENCE EVALUATION AT A LONG-TERM FOOD PROCESSING LAND TREATMENT SITE

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PERFORMANCE EVALUATION AT A LONG-TERM FOOD PROCESSING LAND TREATMENT SITE

by

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16. ABSTRACT

The objective of this project was to determine the performance of a fullscale, operating overland flow land (OFL) treatment system treating nonhazardous waste. Performance was evaluated in terms of treatment of the applied waste and the environmental impact of the system, particularly surface and ground water quality as well as soils accumulation. The major conclusions were: The long-term operation and performance data indicated that the OLF system consistently achieved a very high level of treatment and pollutant removal, from a surface discharge standpoint. With respect to a control area, located at the OLF site but not subjected to wastewater application or impacts, the accumulation of organic carbon, potassium, zinc, and nickel in the soil at wastewater application areas was evident, as well as the apparent leach of calcium, sodium, sulfate. Neither accumulation nor leaching of chromium, magnesium, and chloride were evident. Although the accumulation of zinc and nickel was evident, the cumulative soil concentrations were well below EPA recommended limits, and several hundred years of continued site use may be expected at present loading rates. Groundwater below the OLF site was moderately saline due to the presence of sodium, calcium, magnesium, chloride, and sulfate. No purgeable or extractable organics were detected. Geochemical data indicates the enhanced dissolution and leaching of naturally present soil minerals is due to the infiltration of large volumes of treated wastewater.

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FOREWORD

EPA is charged by Congress to protect the Nation's land, air, and water systems. Under a mandate of national environmental laws focused on air and water quality, solid waste management, and the control of toxic substances, pesticides, noise, and radiation, the Agency strives to formulate and implement actions which lead to a compatible balance between human activities and the ability of natural systems to support and nurture life.

The Robert S. Kerr Environmental Research Laboratory is the Agency's center of expertise for investigation of the soil and subsurface environment. Personnel at the Laboratory are responsible for management of research programs to: (a) determine the fate, transport and transformation rates of pollutants in the soil, the unsaturated and saturated zones of the subsurface environment; (b) define the processes to be used in characterizing the soil and subsurface environment as a receptor of pollutants; (c) develop techniques for predicting the effects of pollutants on ground water, soil, and indigenous organisms; and (d) define and demonstrate the applicability and limitations of using natural processes, indigenous to the soil and subsurface environment, for the protection of this resource.

The Resource Conservation and Recovery Act (RCRA) amendments of 1984 requires the U.S. Environmental Protection Agency (EPA) to assess the adequacy of current federal programs to protect human health and the environment from mismanagement of non-hazardous and unlisted hazardous wastes, which were developed under Subtitle D. The major objective of this project was to determine the performance of a full-scale operating land treatment system treating non-hazardous waste. Performance was to be evaluated in terms of treatment of the applied waste and the environmental impact of the system, particularly, surface and groundwater quality as well as soils accumulation

Clinton W. Hall

Clinton W. Hall
Director
Robert S. Kerr Environmental Research Laboratory

ABSTRACT

The objective of this project was to determine the performance of a full-scale, operating overland flow land (OLF) treatment system treating non-hazardous waste. Performance was evaluated in terms of treatment of the applied waste and the environmental impact of the system, particularly surface and groundwater quality. Performance data were obtained from the Campbell Soup (Texas), Inc. OLF system in Paris TX, which has been in operation for over 24-years. Field samples of soil, wastewater, OLF runoff, and groundwater collected during the project and long-term process records maintained by Campbell Soup were used as part of the evaluation.

The major conclusions were:

1. The long-term operation and performance data indicated that the OLF system consistently achieved a very high level of treatment and pollutant removal, from a surface discharge standpoint. Mass and concentration removals of BOD_5, COD, TOC, and TSS have been consistently high with mean individual pollutant removals >92% (mass basis) and >93% (concentration basis). Total nitrogen removals were between 84 and 89%. Effluent mass discharges have remained well within the regulatory limitations for suspended solids and BOD_5 over the 24 years that the site has been in operation.
2. With respect to a control area, located at the OLF site but not subjected to wastewater application or impacts, the accumulation of organic carbon, potassium, zinc, and nickel in the soil at wastewater application areas was evident, as well as the apparent leach of calcium, sodium, sulfate. Neither accumulation nor leaching of chromium, magnesium, and chloride were evident.
3. Although the accumulation of zinc and nickel was evident, the cumulative soil concentrations (200 kg/ha and 85 kg/ha) were well below EPA recommended limits (1,120 kg/ha and 560 kg/ha) and several hundred years of continued site use may be expected at present average loading rates (3 kg/d and 0.2 kg/d).

4. The OLF site is underlain by several meters of a heavy inorganic clay (Bonham Formation). A semi-confined aquifer, partially confined from above by the Bonham Formation clays and slowly recharged by downward leakage through the Bonham Formation exits below the OLF site. The aquifer is confined from below by the dense, fissile shale of the Eagle Ford Group.
5. Groundwater below the OLF site was moderately saline due to the presence of sodium, calcium, magnesium, chloride, and sulfate. No purgeable or extractable organics were detected.
6. Ionic ratios of major ions in groundwater collected from lysimeters at the OLF site in 1968 exhibited excellent agreement with comparable ionic ratios for samples collected between 1987-1989. The pattern of ionic composition with relatively small changes in ionic ratios suggested a trend toward the dissolution and concentration of naturally present minerals (in the soil) in the relatively slow moving groundwater.
7. Geochemical data indicated that sulfate-chloride facies were dominant for the groundwater collected at 3 monitoring wells at the OLF site and for the lysimeter data collected in 1968. The similarity of ionic ratios and hydrochemical facies with an increase in groundwater constituent concentrations over 20 years strongly suggested a pattern of geochemical evolution due to the enhanced dissolution and leaching of naturally present soil minerals due to the infiltration of large volumes of treated wastewater.

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NOMENCLATURE

Symbol	Description	Dimensions
A	area	L^2
α	statistical level of significance	---
C_i	calculated ionic concentration	M/L^3
C_{mi}	measured ionic concentration	M/L^3
C_s	well configuration shape factor	---
${}^{i}_{j}F$	F-statistic value, i and j degrees of freedom	---
$f(x)$	normal distribution function	---
$i, \frac{dh}{dl}$	hydraulic gradient	L/L
k	hydraulic conductivity	L/T
L	well screen length	L
μ	population mean	---
π	3.14286	---
q	groundwater seepage rate	L^3/T
r	well radius	L
r^2	square of correlation coefficient	---
R	well screen radius	---
σ	population standard deviation	---
S_x	groundwater drawdown	L
t_x	time to drawdown	T
W_{sol}	mass of extraction fluid	M
W_w	mass of soil water	M
x	sample value	---
X	independent variable	---
Y	dependent variable	---

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This project has been an exemplary cooperative effort between: (a) the U.S. Environmental Protection Agency (EPA), particularly individuals at the Robert S. Kerr Environmental Research Laboratory (RSKERL), Ada, Oklahoma; (b) the Campbell Soup Company and; (c) the Environmental and Water Resources Engineering Program of the University of Texas at Austin (EWRE-UT). This report represents the hard work, interest and dedication of the many individuals who have assisted in the project. The contribution of the following individuals deserves specific recognition and is gratefully recognized and greatly appreciated:

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CHAPTER 1

INTRODUCTION

BACKGROUND

The Resource Conservation and Recovery Act (RCRA) amendments of 1984 requires the U.S. Environmental Protection Agency (EPA) to assess the adequacy of current federal programs to protect human health and the environment from mismanagement of non-hazardous and unlisted hazardous wastes. Because the land disposal options of concern for listed hazardous wastes include landfills, land treatment, waste piles, surface impoundments and salt domes, the EPA evaluation of RCRA Subtitle D unlisted wastes includes these options. As a result, information is needed to support EPA activities related to the evaluation of Subtitle D wastes.

The EPA Office of Solid Waste (OSW) has assessed the extent of land disposal for Subtitle D industrial wastes. The assessment indicated that at least 145 million metric tons of industrial non-hazardous waste (35% of all such waste) are managed by on-site landfills, surface impoundments and land application. The OSW assessment also indicated significant missing information, especially data related to the performance achieved when such disposal options are used. Much of the available data were over 10 years old and little information existed on the impact that such operating land disposal options have on human health and the environment.

OBJECTIVES

The major objective of this project was to determine the performance of a full-scale, operating land treatment system treating non-hazardous waste. Performance was to be evaluated in terms of treatment of the applied waste and the environmental impact of the system, particularly surface and groundwater quality. Performance data was to be obtained from a land treatment system that had been operating for an extended period of time. Equally important was the cooperation of a company that: (a) had comprehensive, long-term data about the characteristics of the wastes that have been applied, (b) was able to identify the application rates that have been used and the acreage to which these wastes were applied, and (c) would permit the existing data to be analyzed and additional wastewater, soil, groundwater and vegetative samples to be taken.

FIELD LOCATION AND STUDY AREA

The Campbell Soup (Texas) Inc. OLF treatment system located in Paris, Texas was utilized for all research described in this report. The following provides a brief summary of the Campbell Soup plant and OLF system and the logic why this system was chosen for evaluation. A complete description of the plant and system is presented in Chapter 4.

The plant processes about one billion cans of soup per year and employs about 1,600 people. The site has about 364 hectares (700 ac*) that are being used for the land treatment of vegetable processing wastewater by the OLF method. About 16,000 cubic meters/day (4.25 mgd) are handled in this manner. Only the vegetable processing and can preparation wastewaters are applied to the site. All sanitary wastewaters from the plant are discharged to the Paris, Texas, municipal wastewater treatment plant. In addition, all stormwater and can cooling water are handled separately and discharged to surface streams.

The size of the land treatment system has increased over the years. The original site consisted of 120 ha (300 ac). Three expansions have occurred: each of approximately 80 ha (200 ac). The expansions resulted in a total of 365 ha (900 ac) being included in the OLF system. Not all of the site is available for wastewater treatment; over 485 ha (1,200 ac) comprise the entire site.

The original fields have have been in operation the longest, since the 1960's. The second set of fields began to be used in the early 1970's and in the late 1970's the third set of fields began to be used. This allowed the opportunity to evaluate soils that have received wastes for different time periods.

** Conversion factors relating metric and U.S. units are presented in Chapter 11.*

About 1.5-cm (hydraulic application rate) of wastewater are applied to the wetted acreage each day. Prior to application, the wastewater is screened to remove large solids, and grease is skimmed. The plant and the land treatment site operate all year. No storage is used. The wastewater is pumped directly from a 375-cubic meter sump (100,000-gallon) and is applied by spray headers that have a spray radii of 30-m (150-ft). The overland flow slopes are about 55-m (170-ft) in length.

The soils at site are clays and sandy, clay loams. A semi-confined aquifer exists below the site. Depth to this aquifer varies between 5-m and 10-m (15 to 30-ft).

The vegetation on the land treatment site is Reed canary grass and tall fescue. The crop is harvested two to three times per year and used as forage in the local area.

The logic for choosing this site for this evaluation was as follows. The Paris, Texas site was evaluated in considerable detail in the late 1960's. The early research focused on the surface hydrology and treatment performance of the system. These early studies served as sources of basic information and as "background" data. In addition, many of the Campbell Soup personnel and the RSKERL personnel who were involved in the earlier research were still active and were able to participate in this project. Thus, considerable "historical memory" and knowledge about the plant, available data and the operation of the OLF system was available to the project.

Another important feature was that the initial OLF acreage has been used for over 20 years. Other acreage has been used for shorter periods of time. This allowed the opportunity to evaluate soils that have received wastes for different periods of time.

Most importantly, Campbell Soup agreed to cooperate as actively as possible in the study, to assist with the sampling that was performed and to provide the available historical operational and other data. This included available wastewater characteristic data, effluent monitoring data and weather data. The wastewater data represented the material from the sump and therefore the characteristics actually applied to the site. Such data had been collected approximately once per month for 10-years. In addition, effluent monitoring data have been collected three times a week for 10-years and daily climatological data covering a 15-year period were available.

RESPONSIBILITIES

The project was a cooperative effort between: (a) the Robert S. Kerr Environmental Research Laboratory (RSKERL), U.S. Environmental Protection Agency (EPA), Ada, Oklahoma, (b) the Campbell Soup Company, and (c) the Environmental and Water Resources Program, The University of Texas (EWRE-UT). The general activities of the cooperating organizations were as indicated below.

RSKERL

- o provided technical overview of the project
- o assisted in locating a suitable site for the effort
- o helped plan the field components of the study
- o provided personnel and equipment to collect soil cores, perform a site resistivity survey, install groundwater monitoring wells and collect groundwater samples for analyses.
- o analyzed selected samples for metals, organics and specific non-conventional pollutants.

Campbell Soup Company

After considering possible sites, an excellent one was found. The site is the overland flow (OLF) land treatment facility at the Campbell Soup (Texas), Inc. plant in Paris, Texas. A detailed description of the plant and system is provided in Chapter 4 and a summary of the plant and system is presented later in this chapter. Campbell Soup:

- o provided details about the plant and land treatment system since they were constructed in 1964.
- o provided historical data on the characteristics of the effluent from the land treatment system
- o provided historical characteristics of the wastewater applied to the land treatment system
- o provided estimates of crop yields over the years
- o provided precipitation data collected at the site
- o assisted in numerous aspects of the field work conducted during the study

EWRE-UT

- o obtained the field samples of soil, groundwater, wastewater, and runoff that were part of the project
- o analyzed these samples for conventional, non-conventional and other pollutants and chemicals
- o statistically analyzed historical data of precipitation, wastewater loading and effluent quality for trends and compared these data to data obtained during the project sampling
- o interpreted available data in terms of performance and environmental impact
- o related performance to initial design and recognized land treatment design relationships
- o obtained information about the regional and local hydrogeology
- o prepared final and interim reports

The cooperative effort worked extremely well. In addition to the activities identified above, participants from the organizations met yearly (April, 1987; June 1988; and June, 1989) to plan the project components, to review data as they became available, to discuss the implications of the data, and to initiate project modifications as needed to enhance the effort. Frequent telephone calls between the participants also occurred to facilitate the project activities and to plan specific efforts. Without this exemplary cooperation by all parties, the project activities and the extent of this report would not have been as comprehensive.

CHAPTER 2

CONCLUSIONS

The major objective of this project was to determine the performance of a full-scale, operating land treatment system treating non-hazardous waste. Performance was evaluated in terms of treatment of the applied waste and environmental impact of the system, particularly surface and groundwater quality. Performance data were obtained from the Campbell Soup (Texas) Inc. OLF system in Paris, Texas. The Paris, Texas land treatment has been in operation for over 24-years. The cooperation of the Campbell Soup Company was instrumental to the success of the project. Campbell Soup (Texas) Inc. provided: (a) long-term data about the characteristics of the wastes that have been applied, (b) records of application rates that have been used and the acreage to which these wastes were applied, and (c) permitted the existing data to be analyzed and additional wastewater, soil, groundwater and vegetative samples to be taken.

RAW WASTE CHARACTERISTICS

1. Over a ten-year period between 1977 and 1988 the main characteristics (BOD_5, COD, chlorides, sulfates and solids mass loadings) of the raw wastewater exhibited normal distributions.
2. Statistical comparison of raw wastewater quality between recent and old data indicated long-term consistency with respect to flow and pollutant concentrations.
3. Individual metal loadings were variable over a 12-month period.

LAND TREATMENT SYSTEM PERFORMANCE

4. Seasonally heavy precipitation (>7.5 cm/month) resulted in small increases in TSS mass discharges (200 kg/d to 500 kg/d) within Smith Creek. These increases were well below regulatory mass loading limitations (values) and no statistically significant correlation ($p \leq 0.05$) was found between the TSS loading and precipitation.

5. Because land treatment slope solids discharges were unaffected by heavy precipitation, it appeared that small increases in TSS loading in Smith Creek during periods of heavy precipitation were due to upstream (off-site) sources.

6. The long-term operation and performance data indicated that the system consistently achieved a very high level of treatment, from a surface discharge standpoint. In-stream concentrations of BOD_5, COD, TOC and TSS indicated that mean removals were greater than 93%. Total nitrogen removals were between 84 and 89%. Effluent mass discharges have remained well within the regulatory limitations for solids and BOD_5 over the past 24 years. Percent removals (mass basis) for BOD_5, COD, TOC and TSS have been consistently high (>92%) over the 24-year life of the site.

SOILS

7. With respect to the control area, accumulation of organic carbon, potassium, zinc and nickel in the soil at the wastewater application areas was evident, as well as leaching of calcium, sodium, and sulfate. Neither accumulation nor leaching of chromium, magnesium, and chloride was evident.

8. Although the accumulation of zinc and nickel was evident, the rate of accumulation was so small that several hundred years of continued site use may be expected at present loading rates.

HYDROGEOLOGY

9. The results of the hydrogeologic investigation indicated that an aquifer exists below the OLF site. The data suggested that the aquifer was semi-confined.

10. The results indicated that the erosional contact between the lower confining unit (Eagle Ford Group) and the aquifer (Bonham Formation within the Austin Group) may serve as a significant transmission zone for groundwater within the relatively impermeable clays of the Bonham Formation. Measured hydraulic conductivity values (average k between 10^{-4} and 10^{-3} cm/s) within the erosional contact zone between the Eagle Ford and Austin Groups were substantially greater than typical values for unweathered marine clay and shale (k between 10^{-11} and 10^{-8} cm/s).

11. Groundwater level data indicated that the general flow direction was to the north-west with a hydraulic gradient of approximately 0.004 m/m.

GROUNDWATER QUALITY

12. Twenty six groundwater samples collected from three monitoring wells between 1987 and 1989 were analyzed. The waters were moderately saline (TDS between 7,000 and 13,000 mg/L) due to the presence of Na^+, Ca^{2+}, Mg^{2+}, Cl^-, and SO_4^{2-}. No purgable or extractable organics were detected.

13. Statistical analysis of the groundwater data indicated that, for the major ions present, groundwater quality at each well remained uniform over the three-year period of sampling. Statistically significant differences ($p = 0.05$) in water quality (including calcium, magnesium, manganese, sodium, and sulfate) between the well locations were found.

14. State records indicated that no well casings in Lamar County were screened within the contact zone of the Austin Chalk-Eagle Ford Groups. In addition, no springs which issued from the Bonham Formation were identified within Lamar County or any adjacent counties.

15. The major-ion chemistries (expressed as ionic ratios) of the groundwater samples collected from the confined aquifer were similar at all three monitoring wells. The major-ion composition data suggested the groundwater below the site have undergone similar patterns of geochemical evolution.

16. Ionic ratios of major ions in groundwater data collected from lysimeters at the site in 1968 closely agreed with comparable ionic ratios for more recent samples (1987-1988). The pattern of increasing ionic compostion with relatively small changes in ionic ratios strongly suggested a trend towards the concentration of dissolved minerals in the relatively slow moving groundwater.

17. The geochemical data indicated that sulfate-chloride facies were dominant for groundwater at all three monitoring wells (MW-1, MW-2 and MW-3) and for the lysimeter

data collected in 1968. Thus, the similarity of water quality among well locations as well as the similarity in the pattern of geochemical evolution over the past 24-years were strongly suggested.

18. The moderately saline aquifer below the OLF site was apparently the result of the enhanced leaching of naturally present, soluble soil minerals due to the infiltration of large volumes (based on estimated percolation rates) of treated wastewater over the past 24-years.

CHAPTER 3

GENERAL MATERIALS AND METHODS

INTRODUCTION

Samples of soil, groundwater, effluent and raw wastewater were collected at the Paris, Texas site. The data were used in the evaluation of the long-term performance of the OLF system and its effects on the environment.

Table 3-1 is a summary of the analyses performed and the media (soil, groundwater, wastewater) from which the samples were collected.

TABLE 3-1. SUMMARY OF SAMPLE ANALYSES PERFORMED

Analysis	Soil	Groundwater	Wastewater
COD or %OC	X		X
BOD_5		X	X
TOC		X	X
NH_3-N		X	X
NO_3-N		X	X
NO_2-N		X	X
Chloride	X	X	X
Sulfate	X	X	X
TSS			X
VSS			X
TDS		X	
pH	X	X	X
Conductivity	X	X	X
Alkalinity		X	
Dissolved O_2			X
Metals	X	X	X
CEC	X		
Volatile organics		X	
Base/neutral extractables		X	

ANALYTICAL METHODS

Dissolved Oxygen (DO)

Dissolved oxygen (DO) levels in natural and wastewaters depend on the physical, chemical, and biochemical activities in the water body. The membrane electrode procedure(1) was used for all DO analyses. A YSI Model 58 DO meter and stirring electrode were used for the measurements. The probe was calibrated daily according to the manufacturers instructions.

Biochemical Oxygen Demand (BOD)

The five-day biochemical oxygen demand (BOD_5) determination is an empirical test in which standardized laboratory procedures are used to determine the relative oxygen requirements of wastewaters and effluents. The test measures the oxygen required for the biochemical degradation of organic material (carbonaceous demand) and the oxygen used to oxidize inorganic matter such as sulfides and ferrous iron.

Method 507(1) was followed for the determination of BOD_5 of samples collected at the Paris, Texas site. Samples were collected in 1-L plastic cubitaners and stored on ice until analysis. The maximum storage time for the samples prior to analysis (48-hrs) was within EPA regulatory limits(1). Samples were not seeded. However, nitrification inhibitor (Nitrification Inhibitor 2533, Hach Chemical Co.) was added.

Each sample was prepared at three different dilutions, with a minimum of two replicates at each dilution. Initial and final dissolved oxygen (DO) concentrations were determined using the membrane electrode technique, method 421F(1). A YSI Model 58 dissolved oxygen meter and stirring electrode were used for the measurements. The probe was calibrated daily according to the manufacturer's instructions.

Chemical Oxygen Demand (COD)

The chemical oxygen demand (COD) is a measure of the oxygen equivalent of the organic matter content of a sample that is susceptible to oxidation by a strong chemical oxidant.

The Twistube COD method was used(2). This method has been shown to be equivalent to the standard COD method(2). The prepared reagents (in twistubes) and block heater were purchased from O.I. Corporation. A Beckman Model Lambda 3 spectrophotometer operated at a wavelength of 430 nm was used for the analysis of the digested samples. Standards within the sample concentration range and blanks were prepared and analyzed in duplicate daily. Standards were prepared from potassium hydrogen phthalate as described in method 508A(1). Each sample was analyzed singly with 20 percent of the daily sample load analyzed in duplicate.

Total Organic Carbon (TOC)

Total organic carbon (TOC) is independent of the oxidation state of the organic matter and does not measure organically bound elements such as nitrogen and hydrogen that can contribute to the oxygen demand measured by BOD$_5$ and COD.

TOC was determined by the combustion-infrared method according to method 505(1). A Beckman model 915B TOC Analyzer was used for the analyses. Standard curves for inorganic and total carbon were prepared daily from appropriate standards. Daily blanks were run for the determination of background correction factors. Individual samples, blanks, and standards were injected a minimum of three times per analysis per channel (TC/IC) and the mean response value was used for the determination of the TOC value.

Metals

Total metals and water soluble metals were determined by inductively coupled argon plasma (ICP) spectroscopy. Soil and water samples analyzed for total metals were extracted and prepared according to SW 846 Method 3010(3) Water soluble soil extracts were prepared in 1:5 (by weight) soil to water (ASTM Type II) mixtures and tumbled overnight, Method 10-2.3.2(4). The 1:5 solution was centrifuged, and the supernatant analyzed directly by ICP.

The majority of the soil and water ICP metal analyses were performed at the RSKERL in Ada, Oklahoma. Additional analyses were performed at UT-Austin. The analytical procedures for each ICP unit were similar and are described below.

A minimum of three standards for each metal were used for each calibration. Operating standards were prepared from laboratory grade 1,000 mg/L standards and diluted to appropriate concentrations with ASTM Type II water. The standards were stored in solutions acidified with nitric acid at a pH of less than 2. Based upon the sample matrix, appropriate internal standards were used. Each soil sample was extracted and analyzed in triplicate. Spikes and blanks of the extracts were run at a minimum of 10% (each) of the daily sample load.

Residue

The term "residue" refers to solid matter suspended or dissolved in water or wastewater. Water and wastewater samples collected at the Paris, Texas site were analyzed for total suspended solids (TSS) and volatile suspended solids (VSS) according to methods 209D and 209E respectively(1). Each sample was analyzed singly with a minimum of 20% of the daily sample load run in duplicate.

Total dissolved solids (TDS) content of groundwater samples was determined according to to method 209C(1). Each sample was run in duplicate.

Alkalinity

Alkalinity of a water is its quantitative capacity to react with a strong acid to designated pH. Alkalinity of many surface and groundwaters is primarily a function of carbonate, bicarbonate, and hydroxide content; however, other substances such as borates and phosphates can affect alkalinity values.

Method 403(1) was followed for the determination of total alkalinity values for groundwater samples. Values were reported as alkalinity, mg $CaCO_3$/L to pH 4.5.

Ammonia

Ammonia was analyzed for according to method 417E(1). An Orion Model 701A digital IONALYZER® and an Orion ammonia-selective electrode were used. Operating standards covering the concentrations of 1,000, 100, 10, 1 and 0.1 mg NH_3-N/L were prepared daily and used to calibrate the electrode prior to each use.

Anions

Water soluble soil extracts (as described in the *Metals* subsection), water and wastewater samples were analyzed for nitrate, nitrite, chloride, sulfate and phosphate by ion chromatography[5]. A Dionex Model 10 ion chromatagraph with standard strength eluent (0.003 M $NaHCO_3$, 0.0024 M Na_2CO_3) was used. Operating standards were prepared weekly and analyzed daily according to the manufacturer's instructions. Duplicate samples were run on 10% of the daily samples.

Soil Organic Carbon

Carbon is the chief element present in soil organic matter, comprising from 48 to 58% of the total weight. Thus, soil organic carbon can be estimated through the measurement of loss upon ignition *550°C* multiplied by a factor of roughly 0.5. However, the proportion of carbon in soil organic matter is highly variable for a range of soils, and thus any constant factor selected is an approximation at best. Therefore, it is most appropriate to analytically determine and report the actual organic carbon content of soil rather than convert from organic matter measurements.

The modified Mebius procedure, method 29-3.5.3.1[4] was used for the determination of soil organic carbon percentage. The Mebius procedure is similar to the COD test in that it involves rapid chemical oxidation through the application of heat and the use of potassium dichromate. Each soil sample was run in duplicate and mean values and standard deviations of the soil organic carbon percentages.were reported.

Purgeable Organics

Gas chromatography/mass spectroscopy, Method 8240[3] was used for the detemination of volatile organic compounds in groundwater samples. The Purge-and-Trap method (Method 5030[3]) was used to purge the volatile components of the groundwater prior to analysis. The volatile components were separated via the gas chromatograph and detected using a mass spectrometer which provided both qualitative and quantitative information. The analyses were performed by EPA at the RSKERL in Ada, Oklahoma.

Base/Neutral Extractables

Base/neutral extractable organic compounds were determined by gas chromatography/mass spectroscopy, Method 8270.(3) The analyses were performed by EPA at the RSKERL in Ada, Oklahoma.

Specific Conductivity and pH

Specific conductance is a numerical expression of the ability of an aqueous solution to carry an electrical current. This ability depends on the presence of ions, their total concentration, mobility, valence, and relative concentrations, and on the temperature of measurement.

A self-contained Lab-Line® Model MC-1, Mark V conductivity meter was used to measure aqueous conductivity according to SW-846 Method 9050(3). Aqueous samples were measured directly without prior preparation. Soil samples were prepared in 1:5 (by weight)I soil to water (ASTM Type II) mixtures and tumbled overnight, method 10-2.3.2(4). The 1:5 solution was centrifuged, the conductivity of the supernatant was measured directly. Duplicates were run on 10% of the daily samples.

pH is a measure of the hydrogen ion concentration of an aqueous sample. The electrometric measurement method, SW 846 Method 9040(3), was used for pH determination. An Orion Model 701A digital IONALYZER® and an Orion pH electrode were used.

Aqueous samples were analyzed directly without prior preparation. Soil samples were prepared in 1:1 (by weight) soil-water pastes, method 10-2.3.2(4), mixed for 30 minutes, and measured directly.

Cation Exchange Capacity (CEC)

The cation exchange capacity (CEC) of a colloidal material is defined as the excess of counter ions in the zone adjacent to the charged surface or layer which can be exchanged for other cations(6). The adsorbed cations, including Na^+, K^+, H^+, Mg^{2+}, Ca^{2+}, and Al^{3+}, are not an intergral part of the colloidal crystal lattice structure and can be replaced or exchanged with other cations in the soil solution. The cation exchange phenomenon is of great importance in the soil, as it affects the retention and release of nutrients and other salts, as well as the flocculation-dispersion processes of soil colloids.

The CEC values were determined according to Method 9081 of SW 846(3). Duplicates were run at the rate of 10% of the daily sample load. The CEC was expressed as the number of milliequivalents of cations that were exchanged in a sample with a dry mass of 100-g.

STATISTICAL ANALYSES

Test for Normality

Many parametric statistical tests conducted during this research were based on the assumption that the universe from which the sample was drawn was normally distributed. When samples have a normal distribution, parametric tests have a lower occurrence of Type II error (rejecting the null hypothesis of equal means when it is, in fact, true) and are thus more sensitive than nonparametric tests, at any given level of significance. In contrast, when the data are not normally distributed, nonparametric tests (distribution-free tests) are preferred over parametric tests, as the latter have a higher probability of Type I error (accepting a false null hypothesis). It was therefore important, when possible, to test the assumption of normality.

Normality was tested by the following method. The sample data were transformed into standard normal distribution values, and the normal distribution values were divided by the mean to obtain normal scores(7). The normal distribution values were given by the probability density function:

$$f(x) = \frac{1}{\sigma\sqrt{2\pi}} \exp\left(-\frac{1}{2}\left(\frac{x-\mu}{\sigma}\right)^2\right) \tag{3-1}$$

where:

π = pi
x = sample value,
σ = standard deviation, and
μ = mean.

The normal values were calculated using equation 3-1 by using sample values and the long-term values of the mean and standard deviation.

The normal scores of the data were computed and then the individual sample values were regressed (linear regression) against these corresponding, calculated normal scores. Thus, the sample values were regressed against the normal score values. The correlation (r) of the normal scores with the sample values was calculated. The correlation coefficient is a measure of the closeness of relationship between two variables--more exactly, the closeness of the linear relationship. The square of r can be described approximately as the estimated proportion of the variance of Y that can be attributed to its linear regression on X, while ($1-r^2$) is the proportion free from X. Thus, at $r = 0.9$ ($r^2 = 0.81$), about 80% of the variation in Y can be attributed to its linear regression on X. In other words, 20% of the variation in Y cannot be explained by the regression on X. For the sample data, large values of the square of the correlation coefficient r^2 indicated that regression was statistically significant and thus, the data distribution closely followed a normal distribution(8).

Student's t-test

The student's t-distribution was used to develop confidence intervals about mean values and test for differences between pairs of data when the data fit a normal distribution, but the population standard deviations were unknown, no matter what the sample size was. For all statistical tests comparisons conducted as part of this research, the probability (p) of a Type I error was specified as $p \leq 0.05$, i.e., there was a 1 in 20 chance of accepting the null hypothesis of equal means, when in fact, the means were different.

Analysis of Variance

The ANOVA test can be described as a statistical method for comparing 2 or more populations or treatments. It can be used for testing hypotheses involving two or more means. The null hypothesis of equal means is tested through application of the F-test. ANOVA was used when simple pairwise analysis techniques (such as the paired t-test) were inadequate due the complexity of the data sets. ANOVA techniques were used to decide whether the differences among sample means were large enough to imply that the corresponding population means were different.

One-way ANOVA

In the application of one-way ANOVA, random samples drawn from different populations were are divided into different groups or treatments based upon specific experimental conditions. The ANOVA test was then performed with the means and variances from each treatment. The test was used to identify differences between means when more than 2 treatments or conditions were specified, with an independent variable held fixed. For example, one-way ANOVA was used to decide whether groundwater quality (i.e. the concentration of individual ions, the fixed variables) varied between 3 monitoring wells (the treatments).

The ANOVA test can be described as a statistical method for comparing two or more populations or treatments. It can be used for testing hypotheses involving two or more means. The null hypothesis of equal means is tested through application of the F-test. For the raw wastewater metals data, each sample collection date was considered as an individual treatment and the mean concentration value (calculated from duplicate samples) of each individual metal ion was from each treatment was tested by ANOVA to determine if the concentration of the metal ion remained constant over time.

When statistical differences were found (as indicated by an F-statistic which exceeded the critical F-statistic) differences between the means were tested using the Q-test method(8).

Two-way ANOVA

The ANOVA2 test was used (as opposed to the one-way ANOVA test) when two discrete, independent variables were involved; for example, location and depth. When applied to the soil data, the ANOVA2 test included data from all sampling locations and all depths. Thus, all data collected were used to draw conclusions regarding soil trends.

The ANOVA2 test can be used to assess the impact of interaction between the two independent variables. The significance of the interaction, with respect to the two independent variables, must be evaluated before conclusions are drawn. The ANOVA2 test is unaffected by significant interaction if at least one of the independent variables is at a random level. However, if both variables are fixed, and interaction is significant, no conclusions can be drawn regarding the significance of the two variables. The decision as to whether a variable is at a fixed or random level is based on practical considerations. For the Paris, TX data, both fixed and random variables were specified in the application of ANOVA2; therefore, it was appropriate to test for signficance between treatments even when interaction effects were significant.

CHAPTER 4

TREATMENT SYSTEM DESCRIPTION

INTRODUCTION

The following section provides a physical description of the Campbell Soup (Texas) Inc. site, its history of construction and operation, an accounting of wastes produced at the plant and the methods for their disposal, and a description of the regulatory monitoring requirements for the discharge from the system.

FACILITY BACKGROUND

The Campbell Soup (Texas) Inc., Paris, Texas plant, located in Lamar County in northeast Texas (Figure 4-1), is a heat process canning facility which produces about one billion cans of soup per year.

Heat processed soups, spaghetti products, V-8 juice, and pork and beans are produced at the plant. Raw material is brought to the facility by road freight and rail. The plant operates year-round, 24-hours per day, with the exception of a 2-week shutdown (for maintenance and cleaning) in early June.

The complete plant site encompasses over 550 hectares; however, only about 40 hectares are actually occupied by the plant and associated grounds and roads. Of the remaining acreage, approximately 364 hectares are used for wastewater application, with the balance left as pasture. A site plan is provided in Figure 4-2.

Approximately 1,600 people are employed at the plant, which operates on a system of three-8 hour work shifts, beginning with the 7 AM to 3 PM shift. The first shift includes 800 people, 500 on the second shift and 300 on the third shift. A separate work crew of 7 people is entirely responsible for the operation and maintenance of the land treatment system. Five employees are assigned to the first shift and one person to each shift thereafter. The activities of this latter group include the monitoring of raw wastewater and effluent quality, sprinkler and pressure line repair, access road maintenance, and pretreatment equipment and pump inspections.

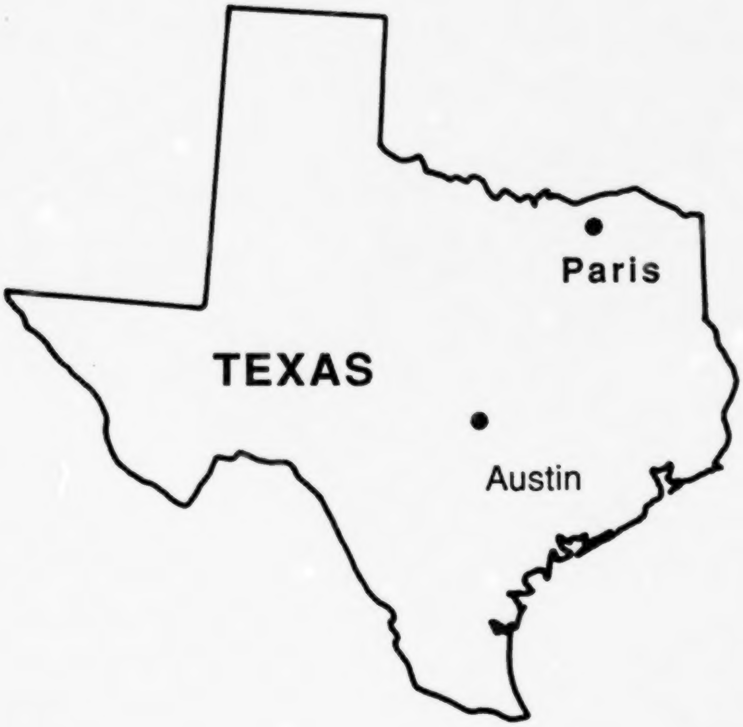


Figure 4-1. General Location Map: Paris,Texas

The Paris, Texas OLF treatment system is located over ground which had been abandoned in the early part of this century due to poor cotton farming management practices which resulted in depletion of soil nutrients and severe erosion[9]. Prior to the initiation of land treatment operations, the land was cleared of trees, brush, grasses and erosion features and graded to form uniform slopes and terraces for treatment purposes.

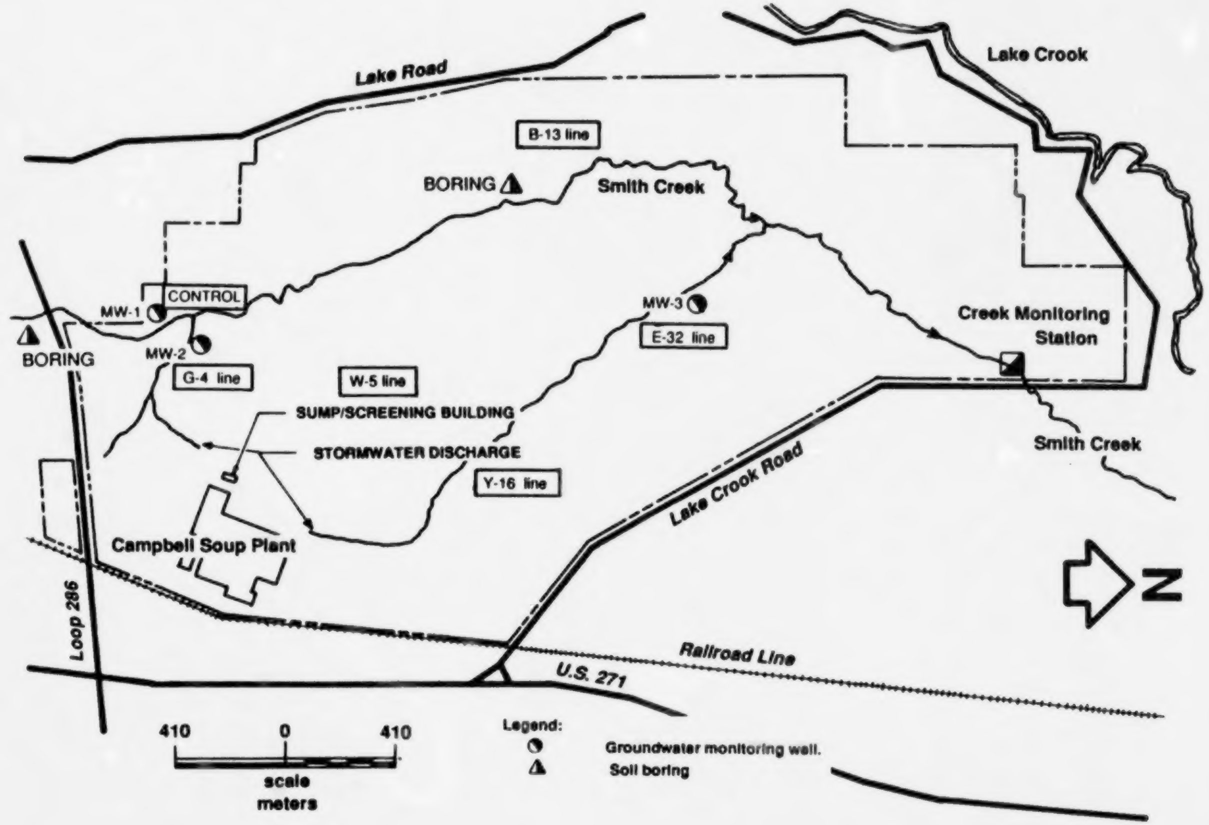

Lake Road
Lake Crook
B-13 line
BORING
Smith Creek
CONTROL
MW-1
MW-2
BORING
G-4 line
W-5 line
MW-3
E-32 line
Creek Monitoring Station
SUMP/SCREENING BUILDING
STORMWATER DISCHARGE
Y-16 line
Smith Creek
Lake Crook Road
Campbell Soup Plant
Loop 286
Railroad Line
U.S. 271
N
410
0
410
scale
meters
Legend:
Groundwater monitoring well.
Soil boring

Figure 4-2. Site Plan: Campbell Soup (Texas), Inc. The terms E-32, Y-16, G-4, B-13, W-5 and CONTROL indicate the locations where detailed site investigations were performed.

Overland flow land treatment operations began at the site in November 1964. At that time, the mean daily wastewater flow was approximatley 16,000 cubic meters/day. In the late 1960's and early 1970's, the Campbell Soup Company anticipated significant expansion of processing operations at the Paris, Texas plant and consequently increased the size of the OLF system. The anticipated processing expansions, however, never occurred and plant output in terms of product and wastewater flow has remained constant. Thus, the addition of OLF capacity (wetted acreage) resulted in progressive reductions in the actual wastewater application rate (volume of wastewater applied per unit area). The effects of the additional treatment capacity on wastewater loading are discussed in Chapter 5.

The overland flow system reached its present size of approximately 365 ha after three successive expansions over the past 25 years. The original application system (constructed in 1964) consisted of 120 ha. Three additions in 1966, 1971, and 1975 (Figure 4-3) of approximately 80 ha each have resulted in the present system.

Soil characteristics at three spray lines (Y-16, G-4, and E-32) were studied in detail. The terminology "spray line" refers to a discrete application area to which wastewater was pumped to through a single distribution pipe (line). The alpha-numeric distribution piping identification system used by Campbell Soup service personel was used throughout this report to maintain continuity with previously published literature. Soil characteristics at a control area which had never received wastewater were also examined.

The Y-16 and G-4 lines were part of the original system and first received wastewater on November 30, 1964. The E-32 line, part of the third expansion in 1975, has received wastewater since 1978. These three spray lines were chosen for detailed study because the differences in their ages provided a means to evaluate the effects of OLF on soil characteristics over time. The B-13 line and the W-5 line were used to assess treatment slope runoff characteristics.

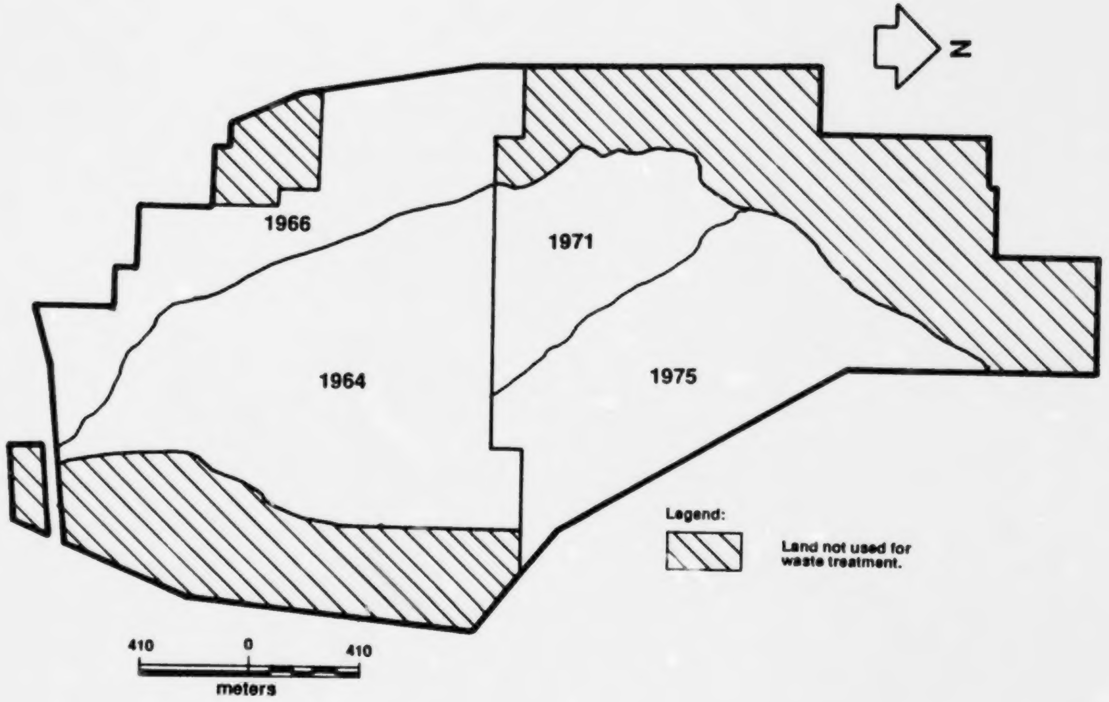

N
1966
1964
1971
1975
Legend:
Land not used for waste treatment.
410
0
410
meters

Figure 4-3. Expansions of the Paris, Texas System

SOLID AND LIQUID WASTE DISPOSAL

Between 1,900 and 3,800 m^3/d of can cooling water is discharged directly into a runoff collection channel which flows into Smith Creek at the south end of the site (Figure 4-2). The cooling water temperature range is between 40 and 60°C. Within the period from January through December of 1987, 44 samples were collected from the cooling water discharge. The mean cooling water quality values were: COD = 40 mg/L, BOD_5 = 6 mg/L, TSS = 10 mg/L. The cooling water input to Smith Creek does not result in a significant degree of dilution of the OLF system runoff because OLF system discharge and cooling water quality are very similar.

Stormwater runoff from plant buildings, parking lots, and associated grounds was discharged into effluent collection channels that flowed directly to Smith Creek (Figure 4-2). Sanitary wastewater produced at the plant was sent directly to the City of Paris municipal treatment system.

All cans were made from sheet metal which was cut and formed at the plant. The cans were made of steel and tin. Some cans were soldered with tin, others were welded. Damaged cans and scrap metal were sold back to the vendor and recycled. Paper labels were printed offsite and glued onto cans at the plant.

The can forming areas were washed down once daily with solvents, detergent and water. The solution was collected and used for energy recovery in a boiler. There was no discharge of this solution into the wastewater treatment system.

Freight loading areas at the rear (north end) of plant were washed down on occasion with detergent and water. The wash water was discharged to the north storm sewer drain (Figure 4-2) and was not released into the wastewater treatment system.

Cooling tower blowdown water was discharged continuously into the north storm sewer drain at the rate of 3.0-m^3/d. The median pH of this wastewater stream was 7.8. Zinc was added to the cooling tower water to maintain a residual concentration of 0.5 mg/L zinc. A 0.05 mg/L free chlorine residual was maintained in the cooling tower water through the addition of HTH (a commercially available biocide).

Inside the processing plant, two separate collection systems directed grease wastewater and vegetable wastewater by gravity to an adjacent building which housed pretreatment, pumping and control equipment.

The grease wastewater traveled through a gravity grease separator in which nonmiscible oils, fats, and grease were skimmed off of the water surface. The grease was temporarily stored on-site in 210-L drums. Approximately 360-L of grease was collected each day. The drums were periodically removed by a local rendering company. The skimmed grease wastewater was combined with the vegetable wastewater and directed through four rotating drum screens (No. 10 mesh) where large vegetable solids were removed and conveyed to a large hopper and dumpster. The accumulated vegetable solids (22,700 kg/d) were hauled to a municipal solid waste landfill 4-5 times per day.

The screened wastewater discharged to a 375-m^3 sump. A mechanical grit lift system periodically removed inert material from the bottom of the sump. Turbulence, due to the short residence time in the sump (approximately one-half hour) and coarse aeration, kept most of the remaining wastewater solids in suspension.

The wastewater level in the sump was automatically maintained by a liquid level controller which progressively activate a maximum of six high-pressure turbine pumps as required. As each pump started, it activated a clock timer. This, in turn, opened the valves in the spray fields that were preprogrammed to that pump. The valves were operated pneumatically. There were four clocks, with three dedicated to 8-hr shifts and the fourth as a variable rate backup mechanism for sump relief when flow exceeded the capacity of a preprogrammed spray line assignment during any one shift.

A flow diagram of the daily disposition of the principal solid and liquid wastewater streams at the Paris, Texas site is provided in Figure 4-4.

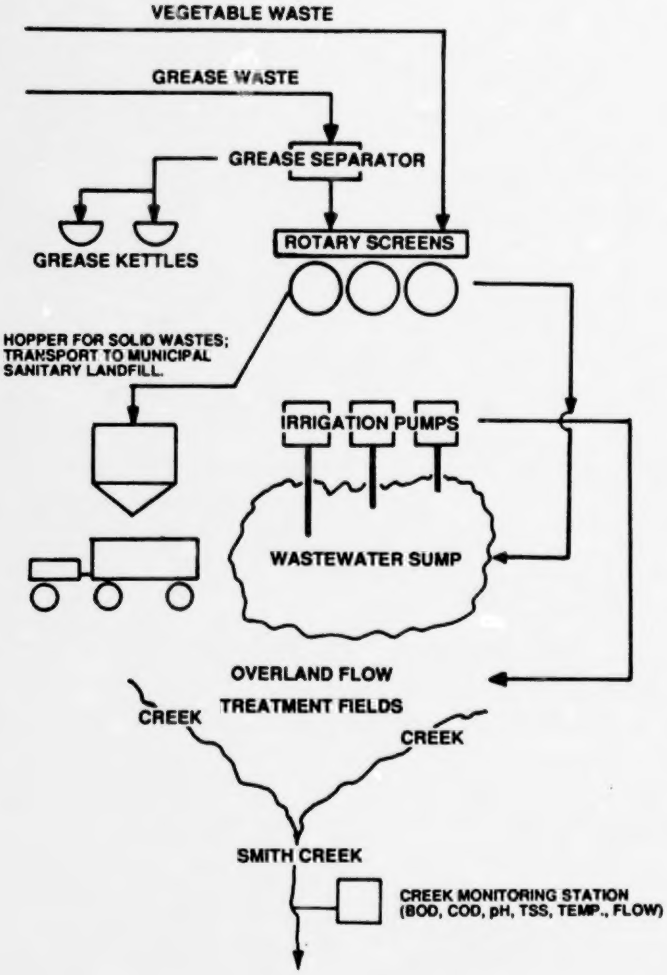

VEGETABLE WASTE
GREASE WASTE
GREASE SEPARATOR
GREASE KETTLES
ROTARY SCREENS
HOPPER FOR SOLID WASTES; TRANSPORT TO MUNICIPAL SANITARY LANDFILL.
IRRIGATION PUMPS
WASTEWATER SUMP
OVERLAND FLOW
TREATMENT FIELDS
CREEK
CREEK
SMITH CREEK
CREEK MONITORING STATION (BOD, COD, pH, TSS, TEMP., FLOW)

Figure 4-4. Flow Diagram of Waste Processing and Land Treatment System

Campbell Soup, Inc. possessed a Texas Water Commission (TWC) Solid Waste Registration Number (# 30453) for the management of solid waste on- and off-site. The registration was not a permit to dispose of waste. The registration number provided access to data stored by the TWC pertaining to solid waste generated at the site. The Paris, Texas plant produced four specific solid waste streams as identified in Table 4-1.

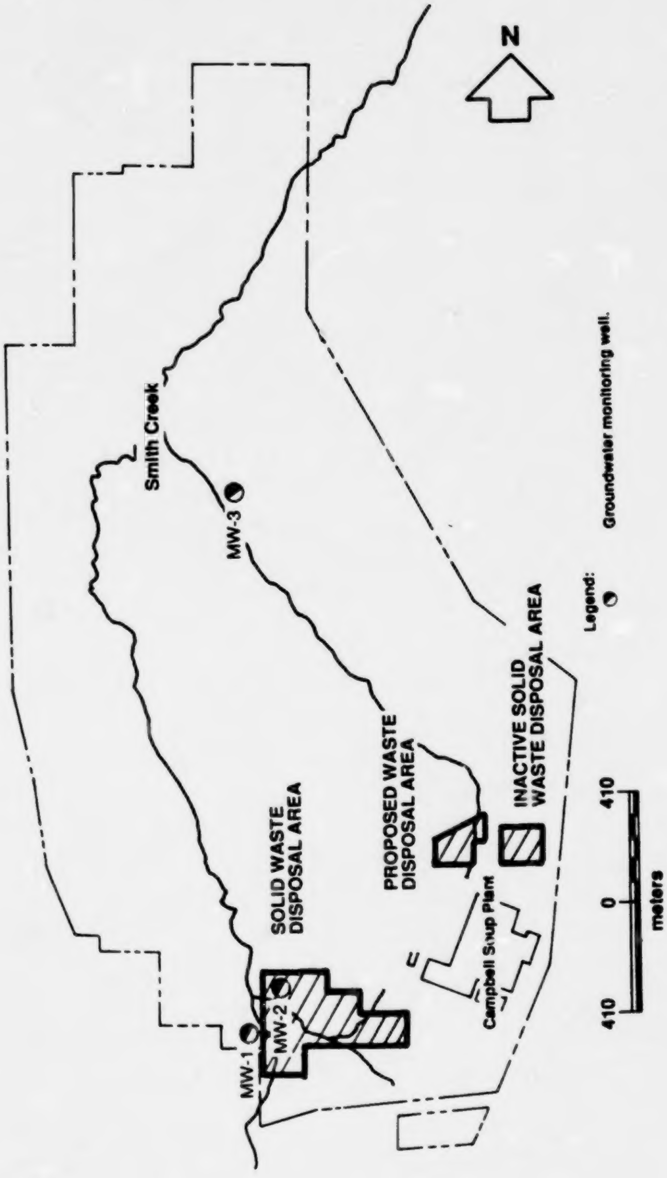
TABLE 4-1. SOLID WASTE IDENTIFICATION AND DISPOSITION

Waste No.	Description	Class	TWC Code	Disposition
1	solvents, spent	I H	910100[+]	on-site/secondary use[1]
2	plant refuse, general misc.	II	279760	off-site[2]
3	food wastes/scrap	II	280010	on-site[3]
4	asbestos insulation	I	179390	off-site[4]

[+] EPA hazardous waste number: F001. Description as per 40 CFR Part 261; Hazardous Waste Management System; Identification and Listing of Hazardous Wastes, U.S.EPA, "The following spent halogenated solvents used in degreasing: tetrachlorethylene, trichloroethylene, methylene chloride, 1,1,1-trichlorethane, carbon tetrachloride, and chlorinated flurocarbons; and sludges from the recovery of these solvents in degreasing operations."

[1] Used for energy recovery in a boiler on-site.
[2] Disposed of in municipal/non-hazardous waste landfill off-site.
[3] Disposed of in unlined trenches on-site.
[4] Disposed of off-site in approved hazardous waste facility.

Waste No. 3 (food wastes/scrap) consisted of "bad product", i.e., canned or bottled soups and sauces which did not meet Campbell Soup's quality standards. These packaged products were buried onsite in shallow trenches (about 3-m deep). Three solid waste disposal (burial) areas for these wastes were located at the Paris, Texas OLF site (Figure 4-5).



N
Smith Creek
MW-1
MW-2
MW-3
SOLID WASTE DISPOSAL AREA
PROPOSED WASTE DISPOSAL AREA
INACTIVE SOLID WASTE DISPOSAL AREA
Campbell Soup Plant
410
0
410
meters
Legend:
Groundwater monitoring well.

Figure 4-5. Solid Waste Disposal Site Locations

SPRAY SITE

In this overland flow (OLF) land treatment process, the screened wastewater was applied to the top portion of graded, vegetated slopes. Treatment occurs as the applied wastewater interacts with the soil, vegetation, and biological surfaces. Physical, chemical, and biological reactions remove most of the contaminants. Treated effluent was collected at the bottom of each slope. The graded clay soils were relatively impermeable and this results in the major portion of the applied wastewater exiting the system in the form of surface runoff. The runoff was collected in ditches or channels at the tail end of the treatment slopes, and was discharged to an adjacent surface water body. Some fraction of the applied wastewater was lost to deep percolation and evapotranspiration. A schematic of a typical installation is illustrated in Figure 4-6.

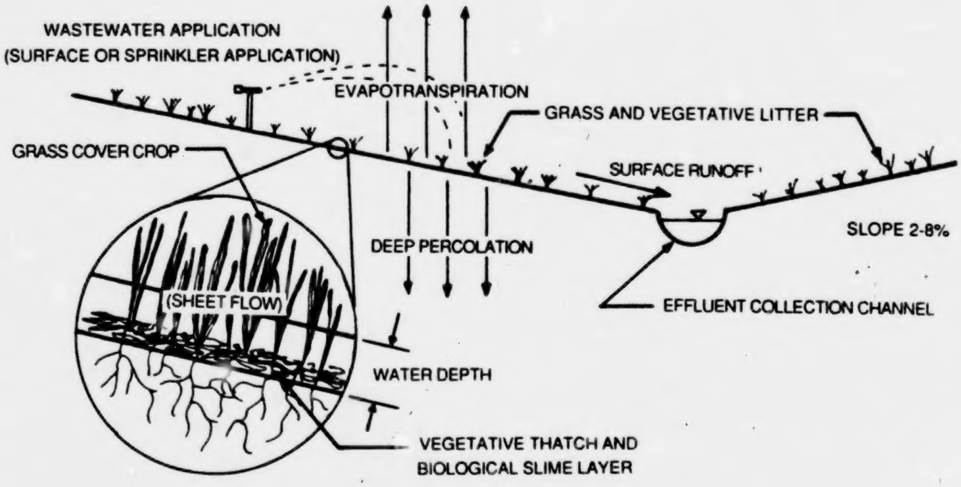

WASTEWATER APPLICATION
(SURFACE OR SPRINKLER APPLICATION)
EVAPOTRANSPIRATION
GRASS AND VEGETATIVE LITTER
GRASS COVER CROP
SURFACE RUNOFF
SLOPE 2-8%
DEEP PERCOLATION
(SHEET FLOW)
EFFLUENT COLLECTION CHANNEL
WATER DEPTH
VEGETATIVE THATCH AND
BIOLOGICAL SLIME LAYER

Figure 4-6. Idealized Overland Flow Slope

On the initial 80 ha at the Paris Texas site, slopes were formed into 1 to 4 ha watersheds following the natural lay of the land, with slope lengths in excess of 75-m. The slopes range from 2 to 8%. Experience gained from the intial years of OLF operation indicated that greater land-use efficiency could be attained without a degradation in treatment performance by forming 0.4 to 1.6 ha watersheds with slope lengths of approximately 55-m. This improved design was used in the three successive expansions.

The wastewater was distributed to the slopes via 40-cm force mains which feed the 10-cm spray line laterals. The lateral lines operate between 270 and 500 kPa and supply the impact sprinklers (80-mm nozzle diameter) which discharge about 76 L/min (maximum rate) each. Each spray line included about 8 to 10 sprinklers, spaced about 21-m apart. The sprinklers had a full circle spray pattern with a 15-m radii. In general, the treatment slopes were 55-m in total length, with the spray header lines located about 15-m downslope from the top of the slope. Thus, the sprinklers were located about 40-m from the collection ditch at the tail end of the treatment slope. This is illustrated in Figure 4-7.

The vegetation on the slopes was reed canary grass and tall fescue, with reed canary being dominant within the spray radii. Outside of the spray pattern, tall fescue, and some coastal Bermuda grass and Dallas grass flourished. These grasses (reed canary and tall fescue) are cool season varieties and are highly water tolerant. Each irrigated hectare yielded approximately 7 to 9 metric tons of hay annually. Typically, the hay was harvested and removed twice a year (July and October) by an outside contractor. However, during unusually wet years, site conditions limited vehicle access and mobility. Under these circumstances, as the weather permited, the hay was shredded on the treatment slopes and left in-place with no baling or removal.

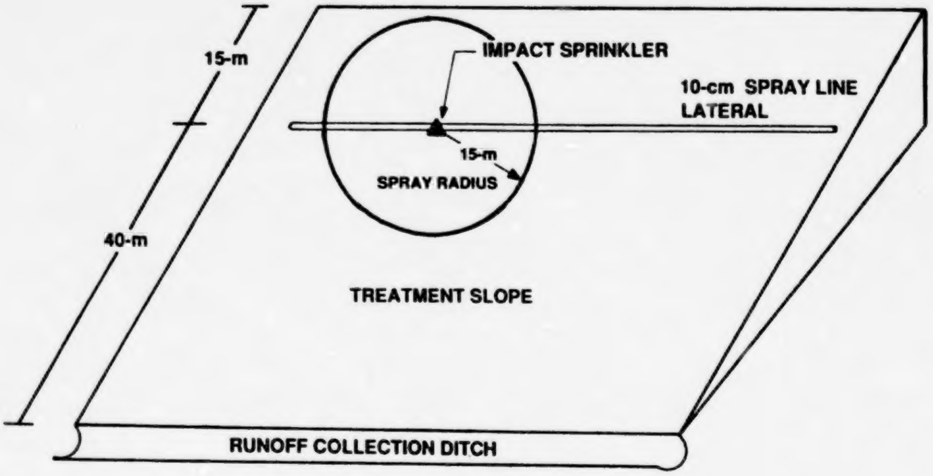


Figure 4-7. Schematic of a Typical OLF Treatment Spray Line and One Sprinkler

Wastewater was applied to each slope for 8-hr per day. Typically, quasi-steady-state discharges (i.e., relatively constant overland flow discharge from the tail end of the slope) occurred between 4 and 8 hours after the initiation of wastewater application. The breadth of this range was due to variation in soil moisture, which differed due to frequency of application, duration of previous applications, climatological variation and transpiration.

FLOW CONTRIBUTIONS TO SMITH CREEK

The treated wastewater (effluent) traveled through a multitude of drainage channels to a central stream (Smith Creek). This creek would be ephemeral without the year-round addition of effluent. During storm events, surface runoff from streets and fields near the north-end of the City of Paris contributed to the flow of Smith Creek. The combined flow of treated wastewater, on- and off-site stormwater discharges, and can cooling water in Smith Creek eventually flowed into the Red River, which corresponds to the Texas/Oklahoma state line (Figure 4-8, General Location Map: Campbell Soup Plant).

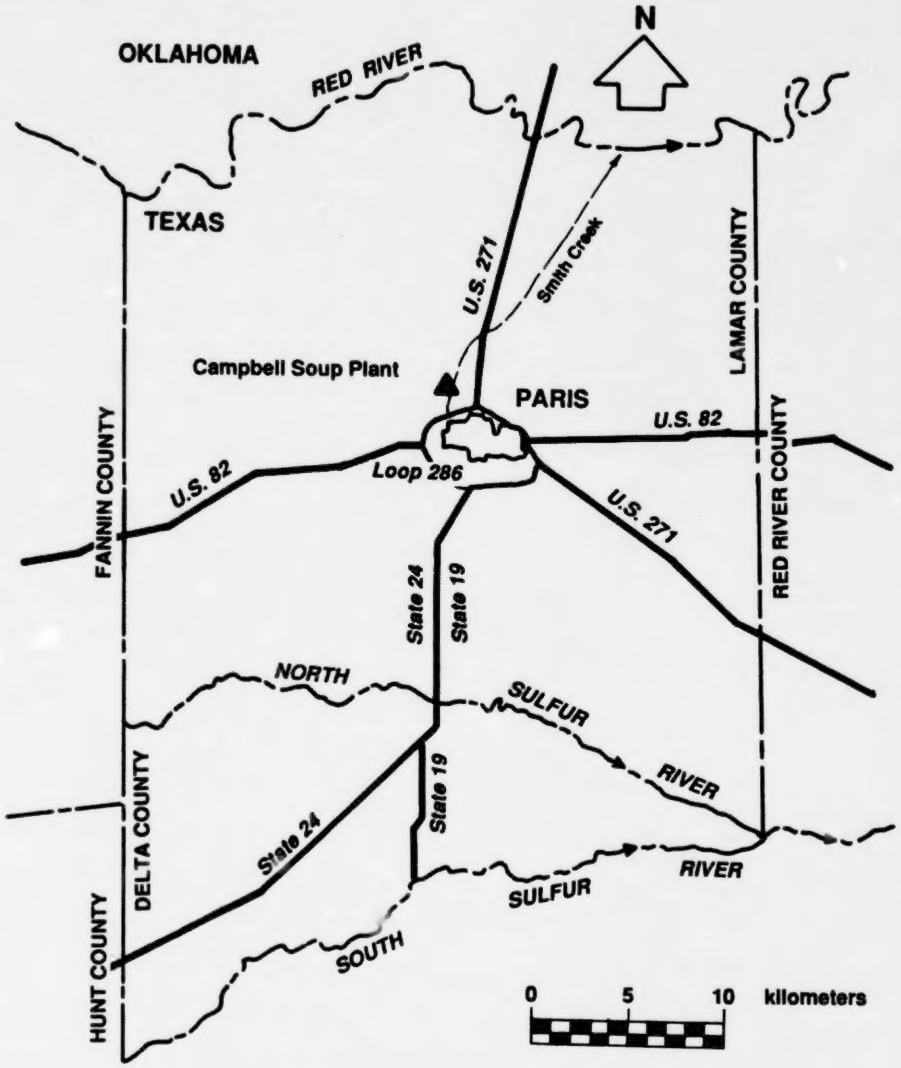

N
OKLAHOMA
RED RIVER
TEXAS
U.S. 271
Smith Creek
LAMAR COUNTY
Campbell Soup Plant
PARIS
U.S. 82
FANNIN COUNTY
U.S. 82
Loop 286
U.S. 271
RED RIVER COUNTY
State 24
State 19
NORTH
SULFUR
RIVER
DELTA COUNTY
State 19
State 24
HUNT COUNTY
SULFUR
RIVER
SOUTH
0
5
10
kilometers

Figure 4-8. General Location Map: Campbell Soup Plant

Water quality of the combined flows in Smith Creek was monitored at an automated sampling station, in accordance with SPDES and NPDES requirements, at the point where the creek exited the property (Figure 4-2). A Parshall flume (maximum capacity 49,205 m^3/d) was used to measure the combined discharge in Smith Creek at this point.

REGULATORY MONITORING REQUIREMENTS

Federal and State regulatory discharge limitations for the combined flows in Smith Creek have been established for BOD_5 and TSS mass discharges and flow. The limitations imposed by the State of Texas and the Federal government were identical with respect to effluent characteristics, discharge limitations, and monitoring requirements. Permit conditions (5-year permits) are presented in Table 4-2.

TABLE 4-2. SMITH CREEK LIMITATIONS AND MONITORING REQUIREMENTS

Effluent Characteristic	Daily Ave. Max. Limit	Daily Max. Limit	Sample Frequency	Sample Type
TSS, kg/d	2,270	3,696	Three/wk	24-hr comp.
BOD_5, kg/d	606	986	Three/wk	24-hr comp.
Flow, m^3/d	37,850	49,205	Continuous	Record
Temperature, °C	32.2	35	Six/day	In Situ

Additional limitations included: (a) the pH of the combined flows at the monitoring station shall not be less than 6.0 nor greater than 9.0, and shall be monitored by grab samples once-per-day; (b) there shall be no discharge of floating solids or visible foam in other than trace amounts; and (c) when the flow of Smith Creek exceeds the monitoring capacity of the Parshall flume, 49,205 m^3/d, no limitations within the permit apply except for the requirement that a BOD_5 of 20 mg/L never be exceeded as measured at the flume.

A comparison between regulatory limitations and actual long-term effluent quality is provided in Chapter 5.

CHAPTER 5

RAW WASTEWATER CHARACTERISTICS

The following section presents long-term raw wastewater characteristics at the Paris, Texas plant. The data were collected by Campbell Soup personnel and UT and EPA researchers. The data were used to establish long-term operating trends and to calculate the land treatment system mass loadings and removal efficiencies.

PREVIOUSLY PUBLISHED DATA

Comprehensive records of raw waste characteristics prior to 1977 were not available. Mean values (from the 1960's) for several parameters of interest were available from early studies at the site. The data are presented in Table 5-1 as published.

TABLE 5-1. RAW WASTEWATER CHARACTERISTICS: MEAN ANNUAL VALUES

Parameter	1973[a]	1968[b]	1968[c]
Flow (m^3/d)	15,140	13,626	11,355
BOD_5	616	492	572
TSS	263	181[d]	245
Total-P	7.6	8.5	7.4
Total-N	17.4	18.8	28.5

All values in units of mg/L, except flow.
[a] Gilde, 1973(9); [b] Thornthwaite, 1969(10); [c] Thomas, et al., 1970(11); [d] VSS.

No information was available as to the statistical confidence of these mean values or the number of samples analyzed and the method of collection. Although 3 separate sources are referenced in Table 5-1, the Thornthwaite report(10) served as the primary source of data for the 2 other reports referenced. The reason for discrepancies between values for identical parameters is not clear, but may lie in each author's method of data summarization. The experimental data collection period ran for 12 months between May 1968 and April 1969. Because these data are the only sources of raw waste characteristics from the late 1960's, they served as a means to evaluate the long-term consistency of the applied wastewater.

UNPUBLISHED PLANT RECORDS

Comprehensive records of raw waste characteristics and flow have been maintained since 1977. These records, covering the period between 1977 and 1988, were made available by Campbell Soup for this research. Composite samples were collected from the sump several times each month (2 to 3 composite samples) and typically were analyzed for BOD_5, COD, oil and grease, TSS, VSS, chloride, and sulfate. Flow, pH, and pumping pressure were continuously monitored at the sump and these data are stored by a microcomputer which also served as the automated control for the wastewaster distribution system. Up-to-date daily and long-term reports could be generated from data stored in the microcomputer. These data have been summarized as part of this project and are presented in this section.

Plots of the cumulative raw data indicated that waste characteristics and mass loading were relatively constant over the 10-year period of record. Application rates (wetted area basis) have varied substantially over the 24-years of operation (Chapter 4); however, the actual mass loadings of principal waste constitiuents have remained constant. It is important to note that application rate is a function of volume applied, duration of application and width of application slope, while mass loading rate is a function of wastewater flow rate and pollutant concentration in the raw wastewater; therefore, the two values are do not necessarily vary congruently.

Water conservation practices instituted at the plant in 1983 apparently resulted in a slight decrease in raw wastewater flow. Although the flow apparently decreased and concentrations of of BOD_5 and TSS remained constant, there was no apparent reduction of BOD_5 and TSS mass loads (Figures 5-1, 5-2 and 5-3). The flow data and BOD_5 and TSS concentration data were tested for normality and for differences between mean values (by one-way ANOVA) before and after the start of water conservation practices in 1983 (Table 5-2). The results indicated that the flow data and BOD_5 and TSS concentration data could be described by a normal distribution. There was no statistical difference in the BOD_5 and TSS concentration data between the 2 time periods tested (1977-1982 and 1983-1988); however, there was a difference in the mean flow values between the 2 time periods.

Table 5-2. ANOVA of Raw Wastewater Flow and Concentrations

Descriptive Statistic	Flow (m^3/d)	BOD_5 (mg/L)	TSS (mg/L)
Normal* Distribution r^2	90.8	87.5	90.2
1977-1982 Mean and Standard Deviation	16,960 ± 1,475	618 ± 357	390 ± 196
1983-1988 Mean and Standard Deviation	14,163 ± 5,000	548 ± 176	416 ± 425
Statistical Difference** Between 1977-1982 and 1983-1988?	YES	NO	NO
Calculated F-statistic	${}^{1}_{56}F = 10.35$	${}^{1}_{78}F = 0.84$	${}^{1}_{78}F = 0.09$
Critical F-statistic	4.01	3.98	3.98

* Results of test for normality (square of correlation coefficient from linear regression of sample values against normal scores).

** One-way ANOVA at $p \leq 0.05$.

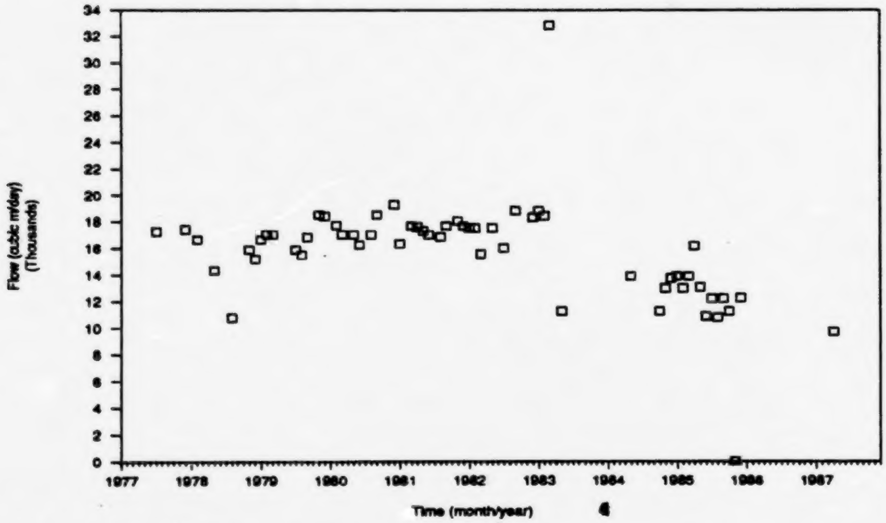


Figure 5-1. Raw Wastewater Flow Rate, 1977-1988

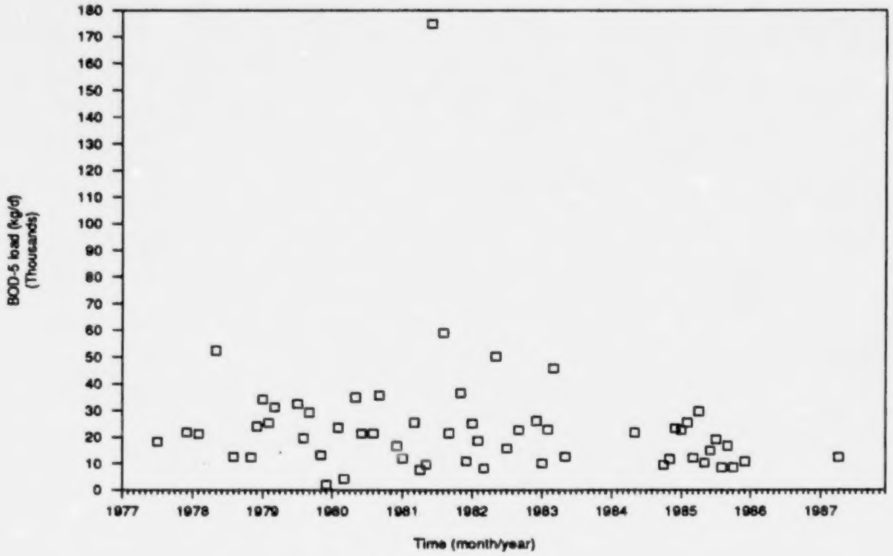


Figure 5-2. BOD_5 Load of Raw Wastewater, 1977-1988

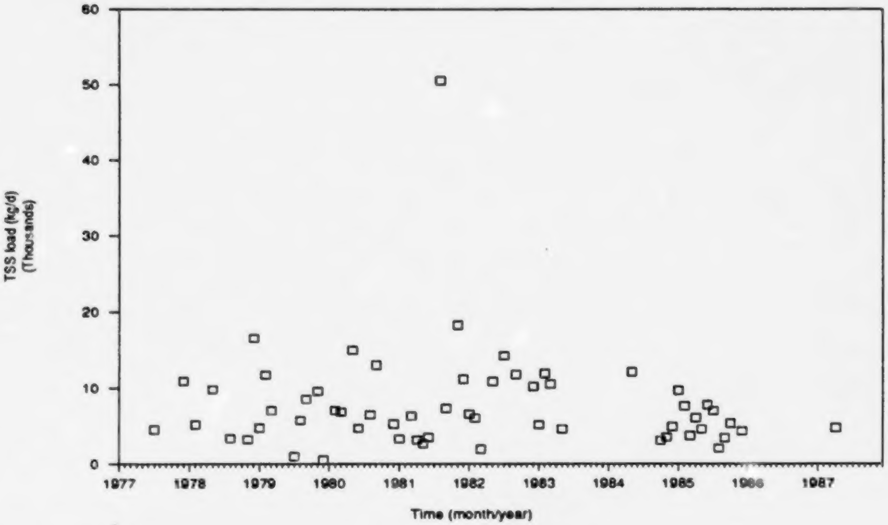


Figure 5-3. TSS Load of Raw Wastewater, 1977-1988

The sump wastewater characteristics for the ten year period between 1977 and 1988 are summarized in Table 5-3. All parameters, with the exception of chloride which was log-normally distributed, fit a normal distribution.

TABLE 5-3. SUMP WASTEWATER CHARACTERISTICS*
PERIOD: 1977-1988

Wastewater Parameters	Mean	Standard Dev.	No. of Data Pts.
Flow (m^3/d)	16,100	3,410	58
BOD_5	550	210	80
COD	1,190	520	78
Oil/Grease	125	80	75
TSS	425	210	80
VSS	370	200	74
Cl^-	66	47	78
SO_4^-	40	20	78
NH_4^+-N	0.7	0.5	11
Org-N	27	9	10
NO_3-N	0.33	0.48	10
NO_2-N	0.03	0.02	10
Total-P	6	3	10

* All values in mg/L unless otherwise noted.

The characteristics of the raw waste remained relatively constant over the 10-year period of data collection, regardless of year (see Figures 5-1 through 5-3), month or season. To illustrate this, mean values and standard deviations for the monthly data from the 10-year period were calculated for flow, BOD, COD, TSS, chlorides, and sulfates. Plots of these representative parameters illustrated the long-term stability of wastewater flow and concentrations (Figure 5-4 through 5-9).

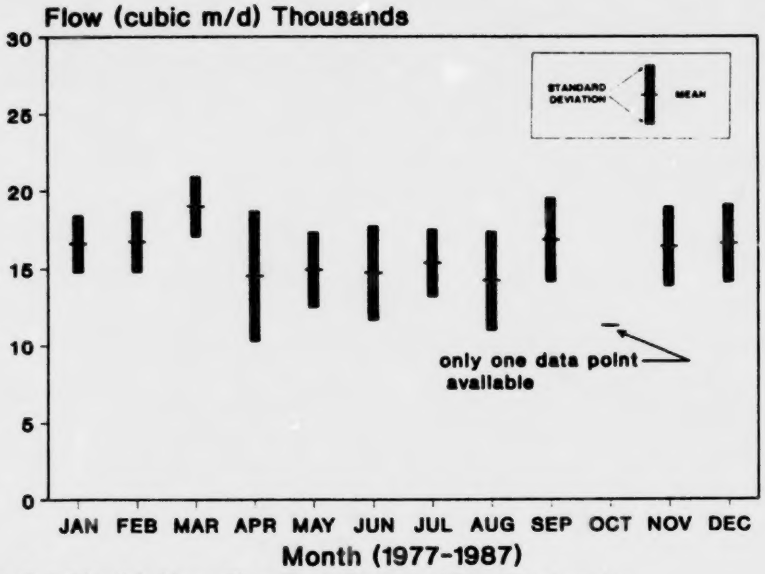


Figure 5-4. Monthly Mean Flow Rate: Screened Raw Wastewater

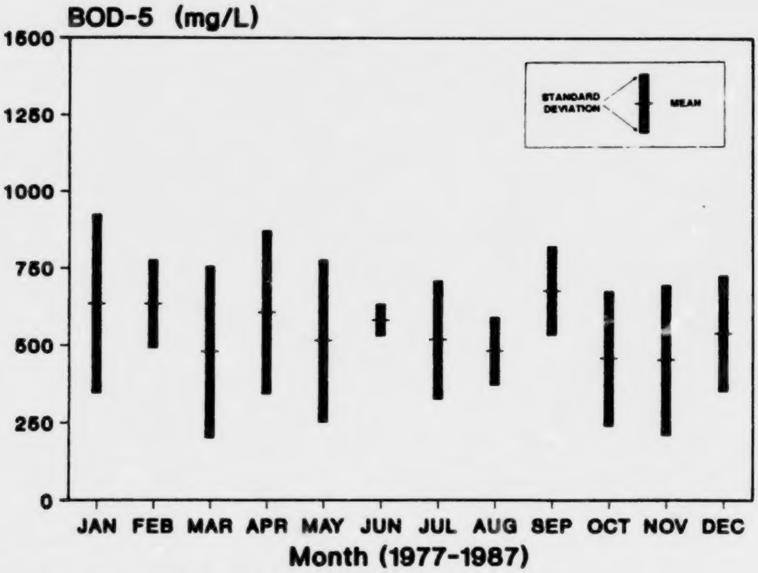


Figure 5-5. Monthly Mean BOD_5 Concentration

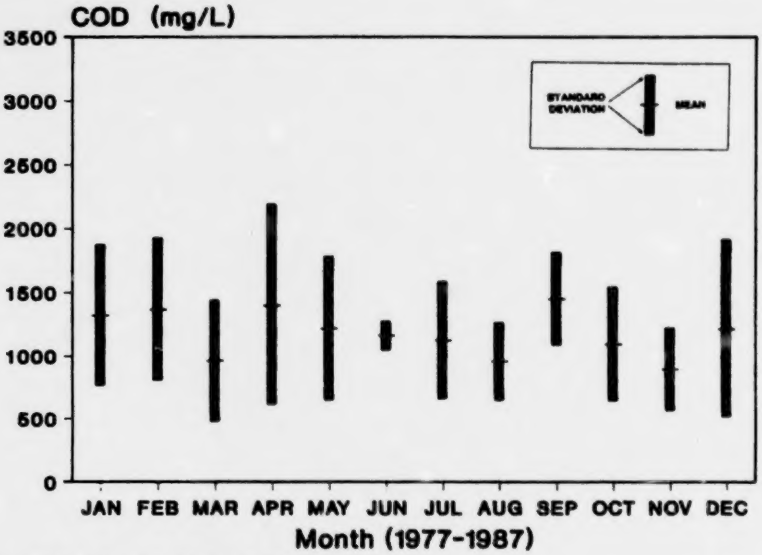


Figure 5-6. Monthly Mean COD Concentration

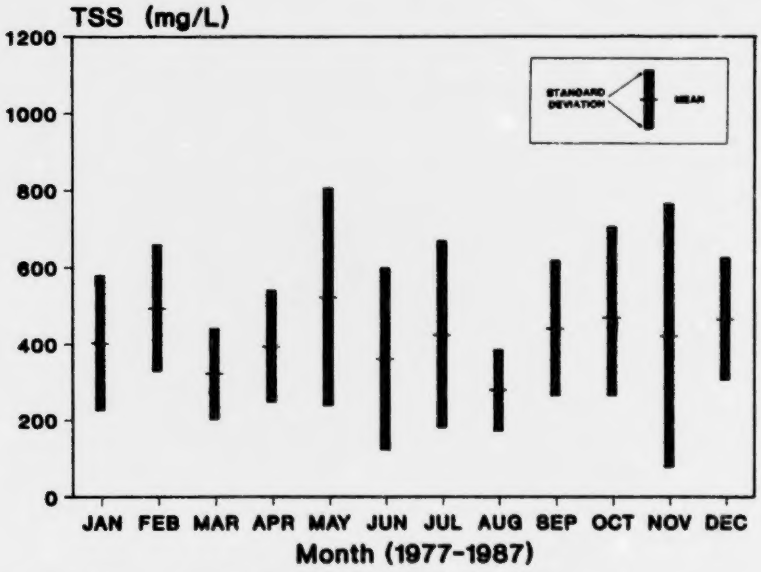


Figure 5-7. Monthly Mean TSS Concentration

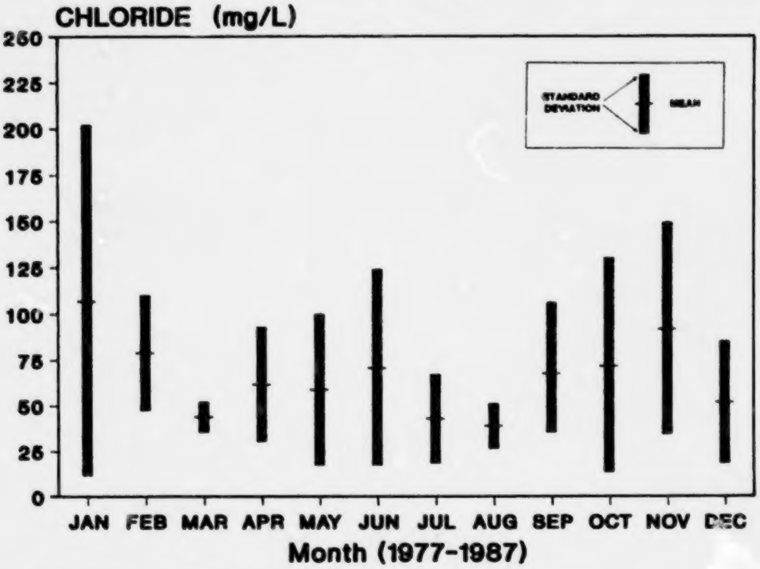


Figure 5-8. Monthly Mean Chloride Concentration

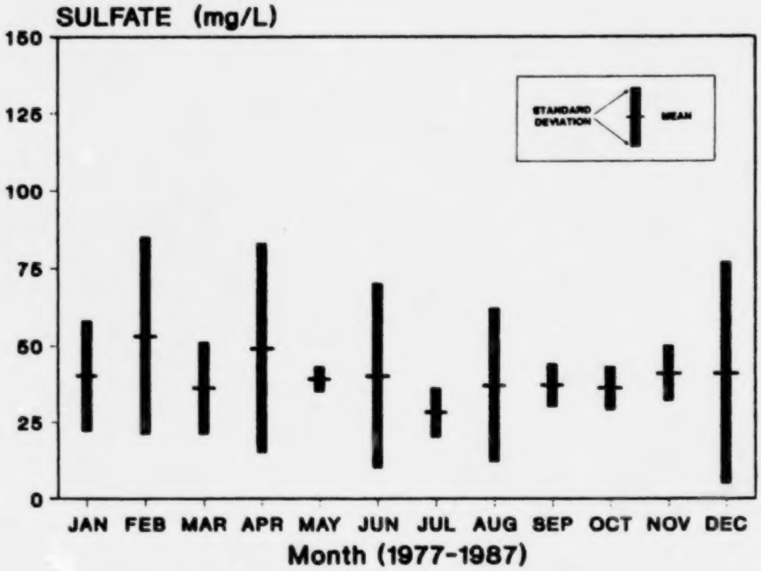


Figure 5-9. Monthly Mean Sulfate Concentration

The monthly summary data did not exhibit any short-term or long-term trends, i.e., monthly or seasonally. This can be attributed to the fact the the plant processes material year-round with essentially no variation in raw material and product quality.

The assessment of the possible accumulation and migration of applied metals performed as part of this study (Chapter 6) required a suitable database of raw wastewater metals concentrations. Because Campbell Soup (Texas), Inc. did not monitor metals in its raw waste, a sampling program was inititated for a period of 10 months (August 88 - May 89). A total of twelve time weighted composite samples were collected by Campbell Soup plant personnel from the screened wastewater in the sump. The acidified samples ($pH < 2$ with HNO_3) were sent to the Robert S. Kerr Environmental Research Laboratory in Ada, Oklahoma for analysis. Cobalt, molybdenum, copper, chromium, nickel, silver, lithium, vanadium, barium, and boron were detected at or below 0.01 mg/l; cadmium and beryllium were not detected at 0.003 mg/l; and arsenic and selenium were not detected at 0.03 mg/L. These values were quantification limits and were determined by instrument (ICP) sensistivity, sample dilution, and matrix interference. A summarization of the data for detected metals is presented in Table 5-4.

TABLE 5-4. TOTAL METALS CONCENTRATIONS IN SCREENED RAW WASTEWATER

Wastewater Parameters	Mean*	Standard Deviation*
Sodium	47.4	15.9
Potassium	24.1	12.1
Calcium	45.1	23.8
Magnesium	4.01	1.36
Iron	0.71	0.41
Manganese	0.04	0.03
Aluminum	0.72	0.63
Zinc	0.17	0.11
Strontium	0.19	0.08

* Concentration units, mg/L.

WASTEWATER MASS LOADING

Wastewater constituent mass loadings were calculated from the long-term data. Only paired data sets (i.e., paired daily flow and concentration data) were used to calculate the loading rates for conventional pollutants (i.e., excluding metals). Because metal concentations and flow data were not paired, mean sump values of metal concentration and wastewater flow (16,100 cubic m/d) were used to estimate metal loading rates. The results are tabulated below (Table 5-5).

TABLE 5-5. WASTEWATER LOADING TO OLF SITE*, PERIOD: 1977-1988

Wastewater Parameters	Mean kg/d	Standard Dev., kg/d	No. of Data Pts.
BOD_5	8,950	4,100	53
COD	19,600	9,555	55
Oil/Grease	2,170	1,345	52
TSS	7,085	3,730	52
VSS	6,330	3,640	51
Cl^-	1,060	780	55
SO_4^{2-}	633	320	56
NH_4^+-N	13	9	7
Org-N	480	178	7
NO_3-N	6	10	7
NO_2-N	0.4	0.2	7
Total-P	91	39	7
Sodium	760	255	12
Potassium	390	195	12
Calcium	725	380	12
Magnesium	65	22	12
Iron	11	7	12
Manganese	1	1	12
Aluminum	12	10	12
Zinc	3	2	12
Strontium	3	1	12

* Based on sump wastewater characteristics and volume pumped.

For the raw wastewater, major parameters of interest, for which sufficient loading rate data were available (i.e., COD, BOD_5, TSS, chloride and sulfate) were tested for normality. The long-term data for BOD_5, COD, TSS, chloride and sulfate exhibited a high degree of agreement with their normal scores. The r^2 values were 0.91, 0.94, 0.96, 0.91, and 0.84, respectively. Therefore, the assumption of normality for the raw waste characteristics was valid.

Because of the demonstrated constancy of the raw wastewater flow and quality, long-term mean values (with associated 95-percent confidence intervals about the mean) have been used throughout the remainder of this report for the calculation of mass balances and land treatment system loadings.

Confidence intervals calculated for the raw waste characteristics during the period between 1977 and 1988 have been summarized and the results are presented in Table 5-6.

TABLE 5-6. CONFIDENCE INTERVALS, SUMP CONCENTRATIONS, PERIOD: 1977-1988

Wastewater Parameters	Mean*	95-Percent Confidence Interval
Flow (m^3/d)	16,100	15,220 - 16,880
BOD_5	550	500 - 600
COD	1,190	1,075 - 1,300
Oil/Grease	125	110 - 145
TSS	425	380 - 470
VSS	370	325 - 415
Chloride	66	55 - 76
Sulfate	40	35 - 45
Ammonia-N	0.7	0.4 - 1.1
Organic-N	27	21 - 34
Nitrate-N	0.33	(see note)**
Nitrite-N	0.03	0.02 - 0.05
Total-P	6	4 - 8

* All values in mg/L unless otherwise noted.

** Reliable confidence interval could be calculated with available data because the standard deviation value was greater than the mean.

The early data (Table 5-1) were compared to more recent data (Table 5-2) and the results are presented in Table 5-7.

TABLE 5-7. COMPARISON OF SELECTED SUMP WASTEWATER CHARACTERISTICS*

Parameter	1973**	1969**	1970**	95% Confidence Interval (1977-1988)
Flow (m^3/d)	15,140	13,626	11,355	15,220 <x< 16,880
BOD$_5$	616	492	572	500 <x< 600
TSS	263	181#	245	380 <x< 470
Total-P	7.6	8.5	7.4	4.1 <x< 7.5
Total-N	17.4	18.8	28.5	NA

* All values in units of mg/L, except flow.
** From Table 5-1.
# VSS
NA Not available. Due to insufficient number of data points.

Comparison of the data indicates that most values from the 1960's data were within or very near the limits of 95% confidence intervals about the mean. Although some values reported in the literature were outside of the 95% confidence interval, the relatively small deviations may be attributed to variation in sample collection location, analytical technique, as well as the unknown size of the 1960's data sets. In general, it appears that waste characteristics have remained relatively constant without significant deviation from the mean over the life of the site. This constancy permited direct comparisons to be made of process performance over the 25-year period of operation.

Confidence intervals were calculated for the raw wastewater metals data (Table 5-8). Also, the metals data were tested by one-way analysis of variance (ANOVA) to determine if there was a statistical significance between the mean individual metal concentrations over time. That is, the ANOVA test was used to determine if the concentration of an individual metal in the raw wasetwater, sodium for example, remained statistically constant over the course of the one year sampling period. For the raw wastewater metals data, each sample collection date was considered as an individual treatment and the mean concentration value (calculated from duplicate samples) of each individual metal ion was from each treatment was tested by ANOVA to determine if the concentration of the metal ion remained constant over time.

The ANOVA testing indicated that at the 5-percent level of significance, other than for calcium, there was statistically significant variation in mean individual metal concentrations in the raw waste over the 12 month study period (Table 5-8). Because the variation in the mean concentration value for each metal was statistically significant, the range of individual values provided by the 95-percent confidence intervals were substituted for mean metal concentrations. The use of concentration ranges provided a statistically sound base for the calculation of individual metal loading rates (Table 5-4) and estimates of probable long-term surface accumulations. These aspects are discussed in detail in Chapter 7.

TABLE 5-8. TOTAL METALS IN SCREENED RAW WASTEWATER

Wastewater Parameters	Mean, mg/L	95-Percent Confidence Interval, mg/L	ANOVA Results*
Sodium	47.4	37.3 - 57.5	$^{11}{}_{2}F = 81$
Potassium	24.1	16.5 - 31.8	$^{11}{}_{2}F = 143$
Calcium	45.1	29.9 - 60.3	$^{11}{}_{2}F = 18.5$
Magnesium	4.01	3.15 - 4.88	$^{11}{}_{2}F = 454$
Iron	0.71	0.45 - 0.97	$^{11}{}_{2}F = 302$
Manganese	0.04	0.03 - 0.06	$^{11}{}_{2}F =$ N/A
Aluminum	0.72	0.32 - 1.12	$^{11}{}_{2}F = 742$
Zinc	0.17	0.10 - 0.23	$^{10}{}_{2}F = 468$
Strontium	0.19	0.13 - 0.25	$^{9}{}_{2}F =$ N/A

* At 5% level of significance, critical F values are: $^{11}{}_{2}F = 19.40$, $^{10}{}_{2}F = 19.39$, and $^{9}{}_{2}F = 19.38$.

N/A Not applicable, ANOVA could not be conducted with available data because the pooled standard deviation values were equal to zero; i.e., for replicate samples, identical concentration values were measured.

Cobalt, Molybdenum, Copper, Chromium, Nickel, Silver, Lithium, Vanadium, Barium, and Boron were detected at or below 0.01 mg/L; Cadmium and Beryllium were not detected at 0.003 mg/l; and Arsenic and Selenium were not detected at 0.03 mg/L.

CONCLUSIONS

The main characteristics (BOD_5, COD, chlorides, sulfates and solids mass loadings) of the raw wastewater exhibited normal distributions over a ten-year period (1977-1988) and this facilitated the use of conventional statistical analysis techniques. Statistical comparison of raw wastewater quality between recent and old data indicated a high degree of long-term consistency with respect to flow and pollutant concentrations.

The results of the metal specific sampling program indicated that metal loadings of individual sampling days were variable over the sampling period. Because of this variability, 95-percent confidence intervals about the mean (concentration basis) were used to calculate loading ranges for individual metals. The calculated ranges for individual metals loading provided a basis for the assessment of soil metal accumulation with respect to applied load. The effect of metal accumulation on site life expectancy is discussed in Chapter 7.

CHAPTER 6

TREATMENT SYSTEM DISCHARGE CHARACTERISTICS

The following section describes the long-term quality of the treated wastewater which discharged from the OLF slopes as well as the quality of the combined flows in Smith Creek.

The performance of overland flow land treatment sites is typically measured by the quality of the runoff or final downstream water. This aspect is discussed in the following section. Performance may also be related to the effect of land treatment operations on other environmental receptors or media, such as soil or groundwater. The performance of the Paris, TX system with respect to these media is evaluated in Chapters 7 and 8.

SMITH CREEK WATER QUALITY

At the monitoring station on Smith Creek (Figure 4-2), water quality and flow rate of the combined flows in Smith Creek were stable with relatively small annual variations over the 15-year period for which records were provided by Campbell Soup. Plots of the in-stream waste constituent concentrations over time are presented in Figures 6-1 through 6-7.

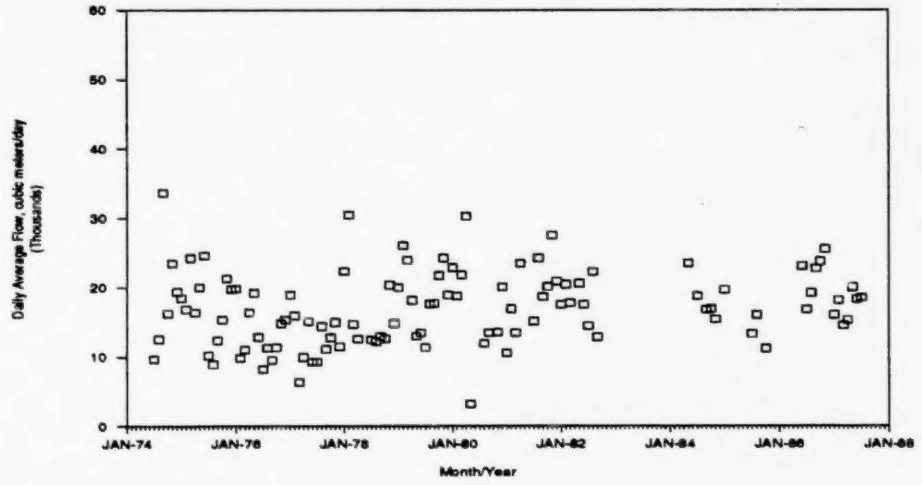


Figure 6-1. Flow Rate of Smith Creek: 1974-1988

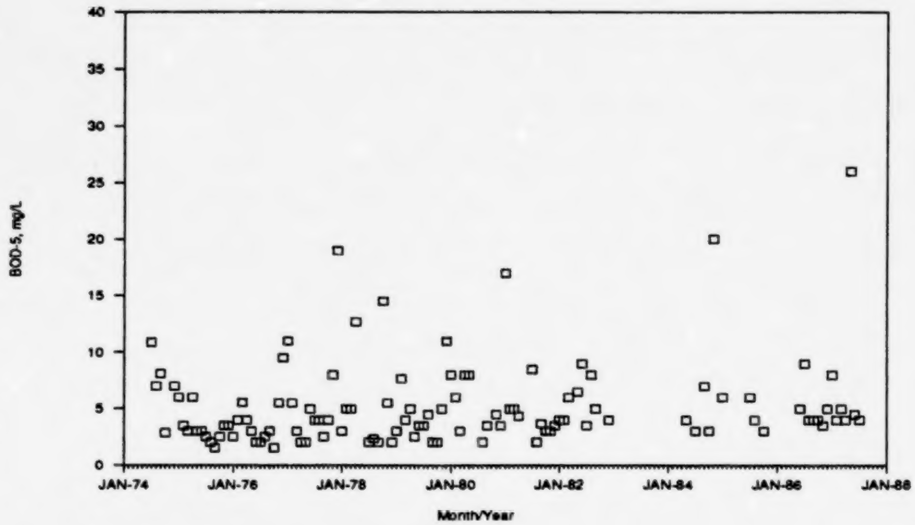


Figure 6-2. BOD-5 Concentration in Smith Creek: 1974-1988

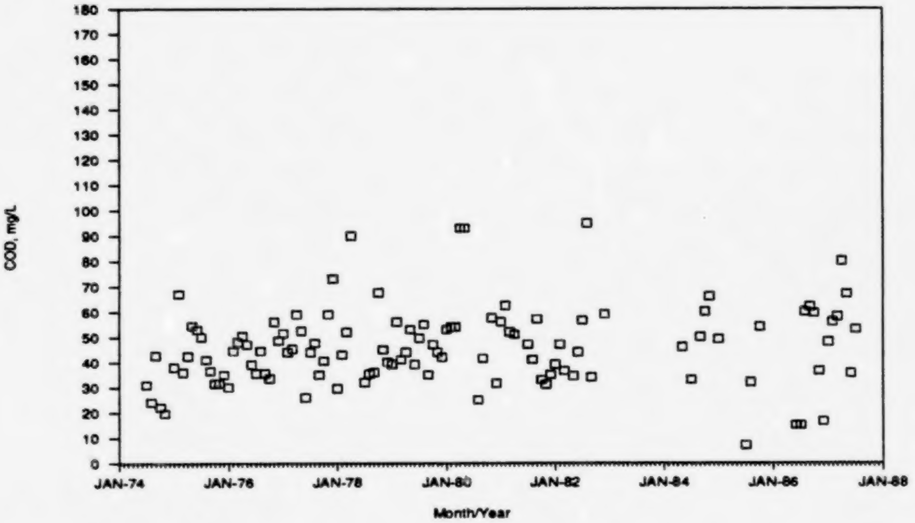


Figure 6-3. COD Concentration in Smith Creek: 1974-1988

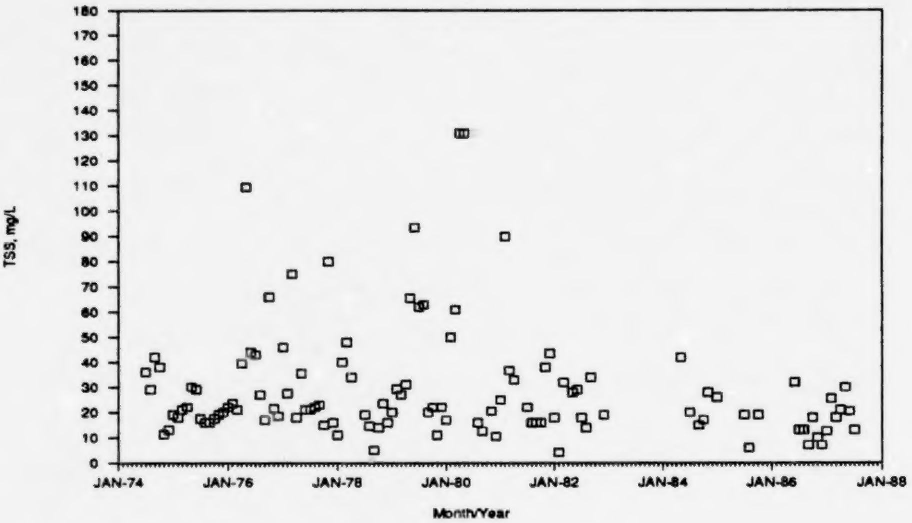


Figure 6-4. TSS Concentration in Smith Creek: 1974-1988

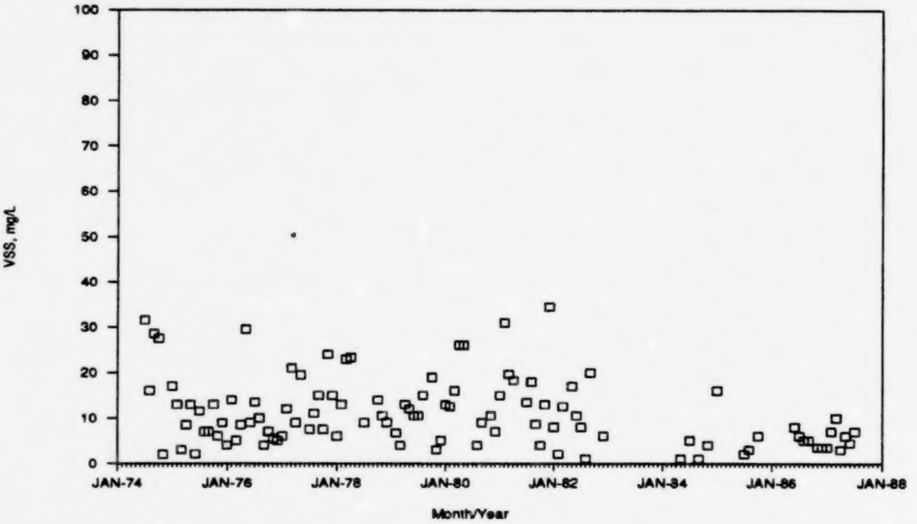


Figure 6-5. VSS Concentration in Smith Creek: 1974-1988

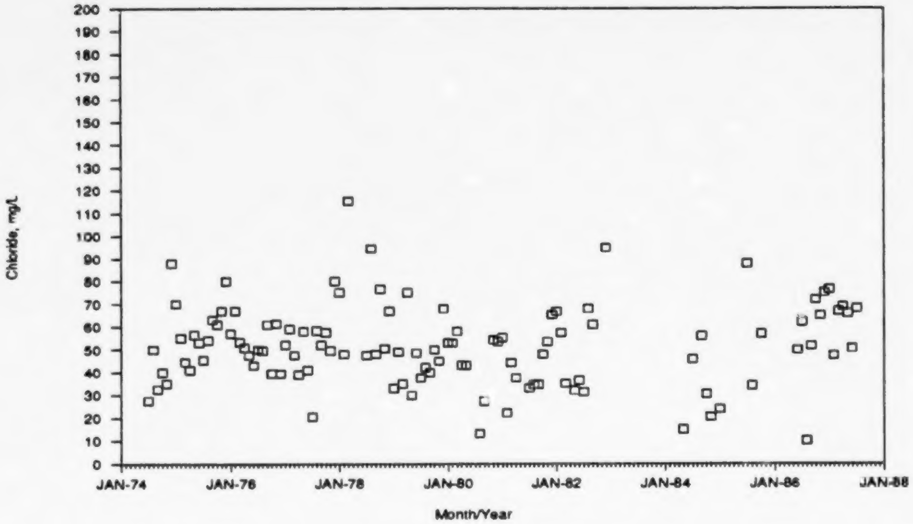


Figure 6-6. Chloride Concentration in Smith Creek: 1974-1988

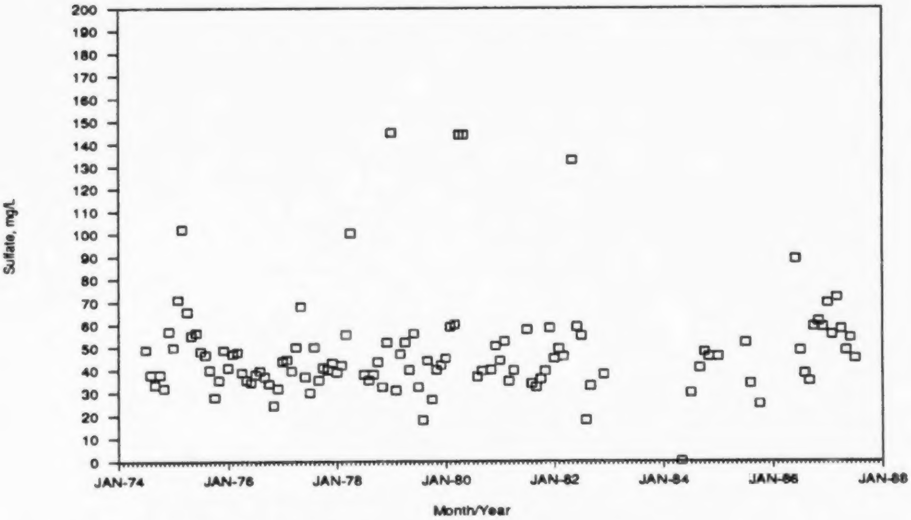


Figure 6-7. Sulfate Concentration in Smith Creek: 1974-1988

A summary, for the period 1974 through 1988, of the collective in-stream concentration data is presented in Table 6-1. The water quality data were obtained from samples collected at the downstream monitoring station, and are representative of the combined flow of Smith Creek, can cooling water, precipitation and the treated wastewater discharges.

TABLE 6-1. CONCENTRATIONS* IN SMITH CREEK: 1974-1988

Parameter	Mean	Standard Deviation	No. of Data Points
Flow (m_3/d)	16,650	5,680	174
BOD_5	5	4	189
COD	46	19	188
Chloride	55	54	188
Sulfate	48	28	186
TSS	33	43	189
VSS	17	42	175

* All values in units of mg/L, except flow.

PERMIT COMPLIANCE

Because Federal and State regulatory limitations have been established for flow, BOD_5 and TSS loads discharged from the Paris, Texas site, an evaluation of the degree of compliance within these limits serves as a convenient and the most common measure of system performance. Discharge limitations governing Smith Creek water quality were listed in Table 4-2.

Plots of the daily average waste loading values (monthly means) over the period between 1974 and 1988 indicated that the quality of the combined discharge in Smith Creek was continuously well below regulatory limits. Plots of BOD_5 and TSS loading illustrate this exemplary permit compliance (Figures 6-8 and 6-9).

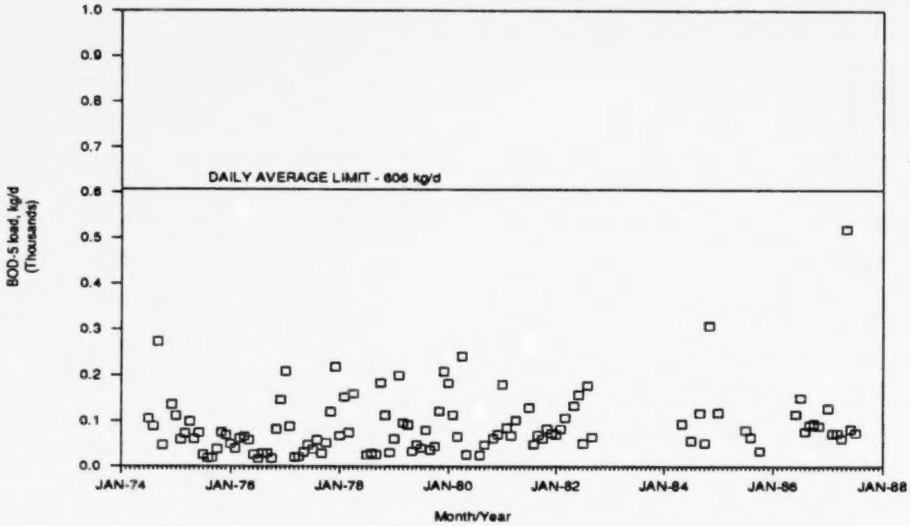


Figure 6-8. BOD-5 Load in Smith Creek: 1974-1988

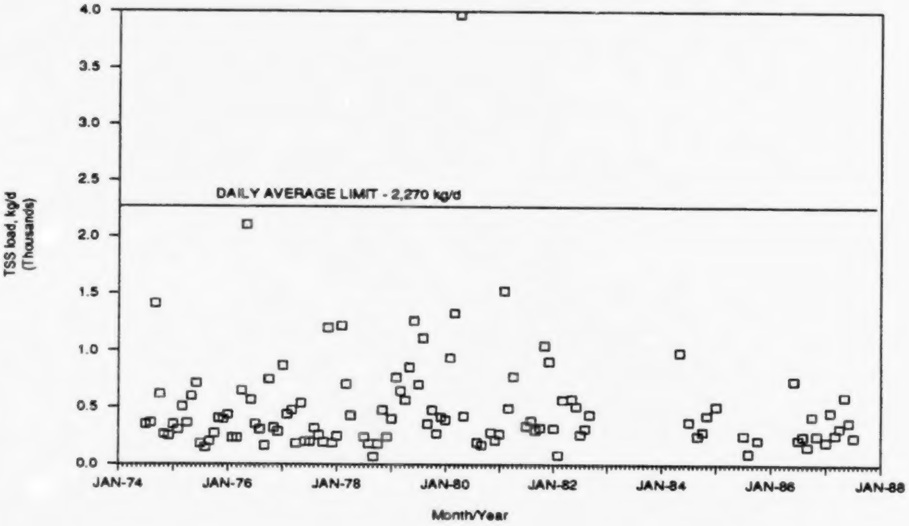


Figure 6-9. TSS Load in Smith Creek: 1974-1988

The calculation of the mean load values (daily average) excluded flood events, i.e., events when the combined flow in the creek exceeded 49,205 m^3/d, the design limit of the in-stream Parshall flume. This method for the calculation of daily averages was established in the NPDES and SPDES permits.

Confidence intervals about the mean load were calculated for BOD_5, TSS, AND VSS. The results are presented in Table 6-2. The long-term data indicate that the quality of the combined flows of wastewater effluent and ephemeral flow in Smith Creek was consistently well below regulatory limits, with little departure from mean values.

TABLE 6-2. EFFLUENT LOADS* IN SMITH CREEK

Parameter	Mean	95% Confidence Interval	Daily Ave. Max. Limit
BOD_5	80	71 <x< 89	606
TSS	445	378 <x< 512	2,270
VSS	190	162 <x< 218	NA

* Units of load, kg/d.

NA Not applicable, no regulatory limit established for VSS.

PRECIPITATION AND ITS EFFECT ON PROCESS PERFORMANCE

The performance of overland flow land treatment systems can be affected by rainfall(12). However, the effects can be small. Studies at indoor pilot and full-scale outdoor facilities at Davis, California indicated that effluent BOD_5 concentrations during rainfall events were only slightly higher than values observed during normal operation without rainfall and were well below 30 mg/L. Effluent suspended solids concentrations at the outdoor Davis, California site during rainfall events were generally greater than 30 mg/L, but solids were mostly inorganic and observed concentrations were generally less than in rainfall from control slopes that did not receive wastewater. The Davis, California studies were conducted during individual storm events. Similar studies were not conducted at the Paris site; however, 15-year monthly mean values for waste loads in Smith Creek and corresponding mean monthly precipitation values have been summarized (Figure 6-10).

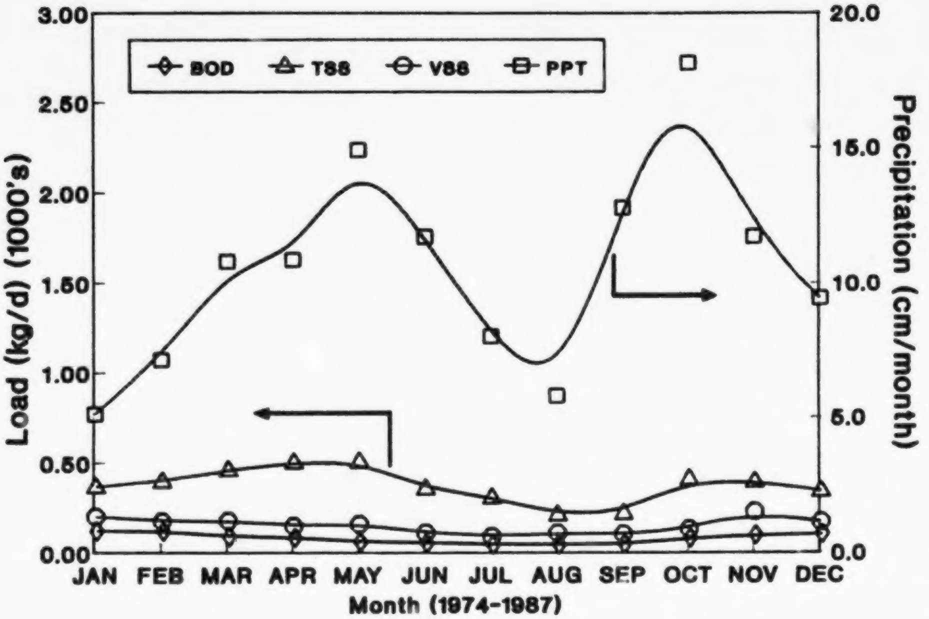


Figure 6-10. 15-year Mean Monthly Waste Loading in Smith Creek and Precipitation

The mean ratio for 81 data points (for the period 1977-1988) of TSS to VSS concentrations in the raw waste was approximately 1.15. Linear regression of TSS against VSS resulted in an r^2 value of 0.92, with the regression equation: TSS,mg/L = 43.2 + (1.01 x VSS,mg/L). A plot of the residuals (observed minus fitted values from the regression equation) against VSS indicates that the residuals were random, indicative of a highly significant correlation between VSS and TSS. Thus, it appeared that in the applied wastewater load, TSS were almost entirely VSS, and that there was a high degree of confidence in the relationship between the two parameters.

In-stream TSS loading patterns appear to have followed precipitation patterns, (Figure 6-10) with little effect on BOD or VSS loadings. However, linear regression analysis of these data indicated that there was no statistically significant linear correlation between TSS loading and precipitation. The correlation coefficient of the regression of TSS load (kg/d) against precipitation (cm/month) was $r^2 = 0.17$. Regression of TSS load against VSS load resulted in an r^2 value of 0.22. Thus, there was no significant linear relationship between TSS and VSS or TSS and precipitation for the in-stream solids data.

Although seasonal increases in precipitation appeared to result in small increases in TSS, the relationship between the two was not statistically significant. Regression of VSS load against precipitation resulted in an r^2 value of 0.02. Thus, VSS loads did not respond to precipitation. In addition, stream flow exhibited no correlation with TSS load, $r^2 = 0.01$.

The small increases in TSS loading during Spring and Fall (Figure 6-10) have been attributed to inorganic material which washed off streets, culverts, and gulleys and into Smith Creek upstream of the land treatment site during these wet seasons. The inorganic material in the urban runoff would then combine with the dilute effluent from the treatment site, resulting in small increases in TSS. However, because the release and in-stream dilution of these mass discharges may have been quite variable, depending on a multitude of possible conditions upstream and at the site, the correlation between the two was low. Because the source of the solids was upstream and the sampling point was several kilometers downstream, "cause and effect" type correlations were low.

Runoff from the treatment slopes, even during heavy storm events did not contain significant quantities of TSS. During a storm event in May, 1989 over 2.5-cm of rain fell within 12 hours. Samples of the slope runoff collected during this storm showed no increase in either TSS or VSS. End-of-slope runoff samples were collected from two slopes at the site (W-5 line and B-13 line, Figure 4-2). Runoff concentrations are listed in Table 6-3.

TABLE 6-3. RUNOFF CONCENTRATIONS[#] DURING WET[+] AND DRY[++] CONDITIONS

Location	TSS	VSS
B-13	3 (6)	3 (4)
W-5	10 (29)	5 (15)

[#] All values in units of mg/L.

[+] Samples collected May 17, 1989 during a storm of 2.5-cm of precipitation in 12-hr period.

[++] Values in parenthesis are samples collected April 18, 1989, no precipitation within 72-hrs of sample collection, values within parentheses.

The long-term data indicate that process performance, as measured by BOD_5, VSS and TSS loads, remained relatively unaffected by preciptiation and seasonal variations.

SYSTEM MASS BALANCES

Mass balances were made around the Paris, Texas land treatment utilizing the upstream and downstream (Smith Creek) flow and concentration data; and raw waste flow and concentration data supplied by Campbell Soup. For the 10-year period for which comprehensive records existed, there were 36 individual days-of-record for which there existed complete data sets (i.e., data set composed of: upstream and downstream flow rate and concentrations, and raw waste flow rate and concentrations) required for the calculation of mass balances.

With the exception of the month of May, each month was represented by at least one complete data set. The data, however, were collected on individual days and thus, may not be representative of monthly conditions. Because no monthly or seasonal trends were observed for either raw wastewater loading or in-stream loading in Smith Creek (see above for discussion), the daily data were used for the mass balance calculations.

Since the actual land treatment site runoff was not measured, an assumption of the applied wastewater that was discharged as runoff was needed. It has been shown that the runoff percentage at the Paris, Texas site varied seasonally, between 44 and 72 percent; with 60% as the mean(10). In addition, variation in application rate, wetted acreage and soil properties could have also affected the runoff percentage. Therefore, it was assumed that a value of 60% was a suitable estimate for the percentage of runoff flow. As a result, the concentration values shown in Table 6-4 are not exact, but are estimates which provide an approximation of the overall system performance.

The calculated values are presented in Table 6-4, along with measured values (samples collected from the runoff collection ditches) reported by Gilde(9).

TABLE 6-4. MASS BALANCE RESULTS FOR THE PARIS, TEXAS OLF SITE

Parameters	**BOD_5**	**COD**	**TOC**	**TSS**	**T-N**
Runoff concentration (mg/L)					
1977-1988 *(calculated)*[1]	6	43	15	32	2
1969 *(measured)*[2]	9	NA	23	16	3
Concentration change (%)					
1977-1988[3]	98	96	95	93	89
1969[4]	99	NA	NA	94	84
Mass removed (%)[5]					
1977-1988	99	98	97	94	96
1969	99	NA	NA	98	92

[1] Mean values based on mass balance calculation results.
[2] Measured values reported by Gilde(9).
[3] Mean values based on paired daily data: raw waste and runoff concentrations.
[4] Single values based on single values reported for raw waste and runoff concentrations(7).
[5] Values based on loading rates calculated from flow and concentration data.

The data in Table 6-4 indicated that mass removal and concentration change percentages for all parameters were generally in excess of 90-percent. These removals were at the high end of efficiencies reported by Martel(13) for several other OLF land treatment sites. Martel(13) reported removal efficiencies (concentration change) observed at 8 OLF sites in the US: ammonia removal between 60 and 99%, TSS removal between 87 and 95%, and BOD_5 removal between 90 and 99%.

The data from 1969 show excellent agreement with the estimated values from the 1977-1988 period. Thus, it appears that process performance remained relatively stable and a high degree of efficiency was maintained over the entire 24-year life of the system.

CONCLUSIONS

Seasonally heavy precipitation produced small increases in TSS mass discharges within Smith Creek. These increases, however, were well below regulatory mass loading limitations and no statistically significant correlation ($p \le 0.05$) was found between the TSS loading and precipitation. Since solids discharges from the OLF slopes were unaffected by heavy precipitation, the small increases in TSS loading in Smith Creek may have been due to upstream (off-site) sources. Thus, the small increases in TSS mass discharges were probably not due to the OLF system.

The long-term operation and performance data collected at the Paris, Texas OLF land treatment site indicated that the system consistently achieved a very high level of treatment, from a surface discharge standpoint. In-stream concentrations of BOD_5, COD, TOC and TSS indicated that mean removals were greater than 93%. Total nitrogen removals were between 84 and 89%. Effluent mass discharges remained well within the regulatory limitations for solids and BOD over the past 24 years. Percent removals (mass basis) for BOD_5, COD, TOC and TSS were consistently high (>92%) over the 24-year life of the site.

CHAPTER 7

SOILS

INTRODUCTION

The long-term application of wastewater can impact a variety of soil properties which in turn can affect crop maintenance, quality, and site-life. These properties include soil permeablity, cation exchange capacity, metal content, and organic carbon content. The following section describes the results of research conducted with soil samples collected from the Paris, Texas site between 1987 and 1989. Samples were collected at specific wastewater application areas and at one location on-site which had not been subjected to wastewater application. The soil properties within this "control" area provided a comparative measure for the evaluation of the long-term impacts of wastewater application on soils within the treatment area. In addition, where appropriate data were available, direct comparisons of soil properties were made between recently collected samples and previously published results for the Paris, Texas site.

MATERIALS AND METHODS

Soil Core Sampling

A total of 20, shallow depth (1.5-m) soil cores were collected from the three waste application slopes and the control area (Figure 7-1), that is, 5 cores per collection area. The number of core samples and the collection pattern was selected by UT and EPA researchers to provide soil samples which were repesentative within the time and analytical constraints of the project. Two, deeper soil cores (7.5-m and 5.5-m) were collected from the control area and the G-4 area during the installation of groundwater monitoring wells. These two cores extended from the ground surface to the confined aquifer. Details of the installation, development and sampling of the monitoring wells are given in Section 8.

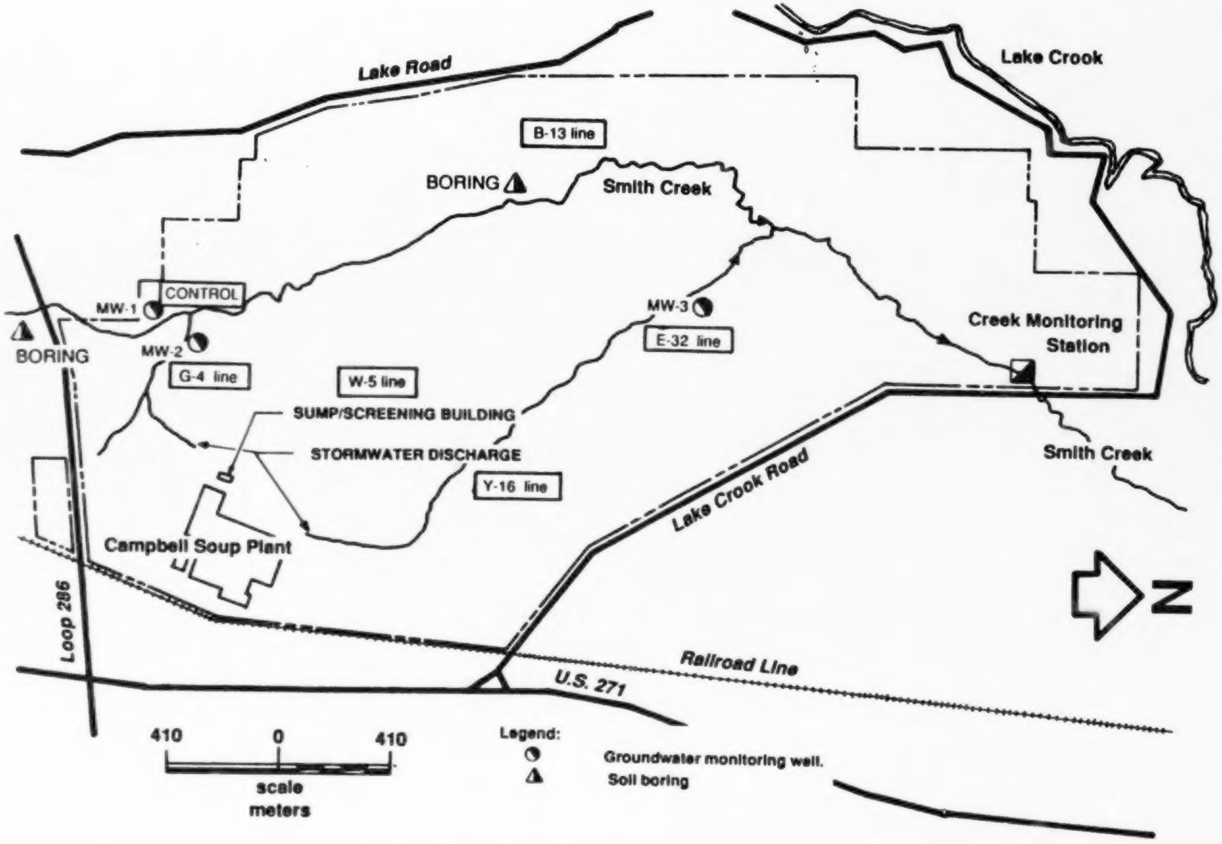

Lake Road
Lake Crook
B-13 line
BORING
Smith Creek
CONTROL
MW-1
MW-2
MW-3
BORING
G-4 line
W-5 line
E-32 line
Creek Monitoring Station
SUMP/SCREENING BUILDING
STORMWATER DISCHARGE
Y-16 line
Smith Creek
Lake Crook Road
Campbell Soup Plant
N
Loop 286
Railroad Line
U.S. 271
410
0
410
scale
meters
Legend:
Groundwater monitoring well.
Soil boring

Figure 7-1. Location Map of Soil Sample Collection Areas

Soil core samples were collected with the aid of a Central Mining Equipment rotating hollow stem auger (10-cm diameter). A core barrel sampler was used to recover undisturbed core material. Five shallow cores were collected from randomly selected locations within the spray pattern at each sample location (i.e., at G-4, Y-16, E-32, and at random from the control area).

After the drilling equipment was readied, the soil surface was lightly scraped to remove any vegetation. A Kimwipe® was placed on the soil surface to identify the true top end of the soil core after removal. The auger bit was lowered on the tissue and soil from the first 15-cm was cored and recovered. The auger bit and core recovery equipment are depicted in Figure 7-2.

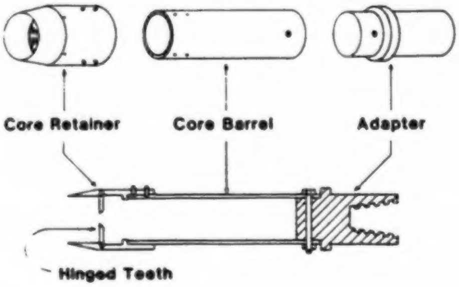


Figure 7-2. Auger Bit and Core Recovery Device

Cores recovered from 15-cm to 150-cm were pared to a diameter of approximately 6-cm. Samples recovered from the top 15-cm were not pared due to the loose, granular structure of the soil. Paring consisted of hydraulically extruding the core through a 6-cm diameter circular paring cone welded to stainless steel plate. The paring apparatus is illustrated in Figure 7-2. The cores were placed in split 6.4-cm diameter PVC pipe. The cores were logged in the field by a geologist.

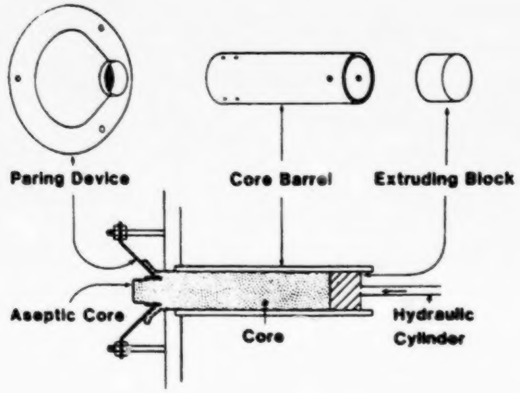


Figure 7-3. Core Extruding and Paring Apparatus

Upon completion of the logging, cores were split in half with a stainless steel laboratory spatula. Each complete core was divided into five sections, 0 to 15-cm, 15 to 30-cm, 30 to 61 cm, 61 to 107-cm, 107 to 152-cm. The selection of these intervals was based upon the expectation that the greatest accumulations of contaminants would be near the surface and therefore, the core sample intervals were smaller near the surface. One-half of each split section was placed in a glass mason jar, the mouths of which were covered with aluminum foil and then sealed with screw caps. Jars were labelled according to sample location and core section depth. Samples were packed on ice in the field, transported to UT-Austin and stored at 4°*C* until analyzed.

Core samples were composited after arrival at UT-Austin. The samples were mixed by hand in a large porcelain bowl with a stainless steel spatula. For each study location (the G-4, E-32, Y-16, and control areas), approximately 5 equal masses of soil from each depth interval from each of the five cores were composited. The compositing process reduced the number of samples scheduled for analysis from 100 to a final total of 20. The original, unmixed core samples were retained and stored at 4°*C*. A flow diagram for the compositing sequence is given in Figure 7-4.

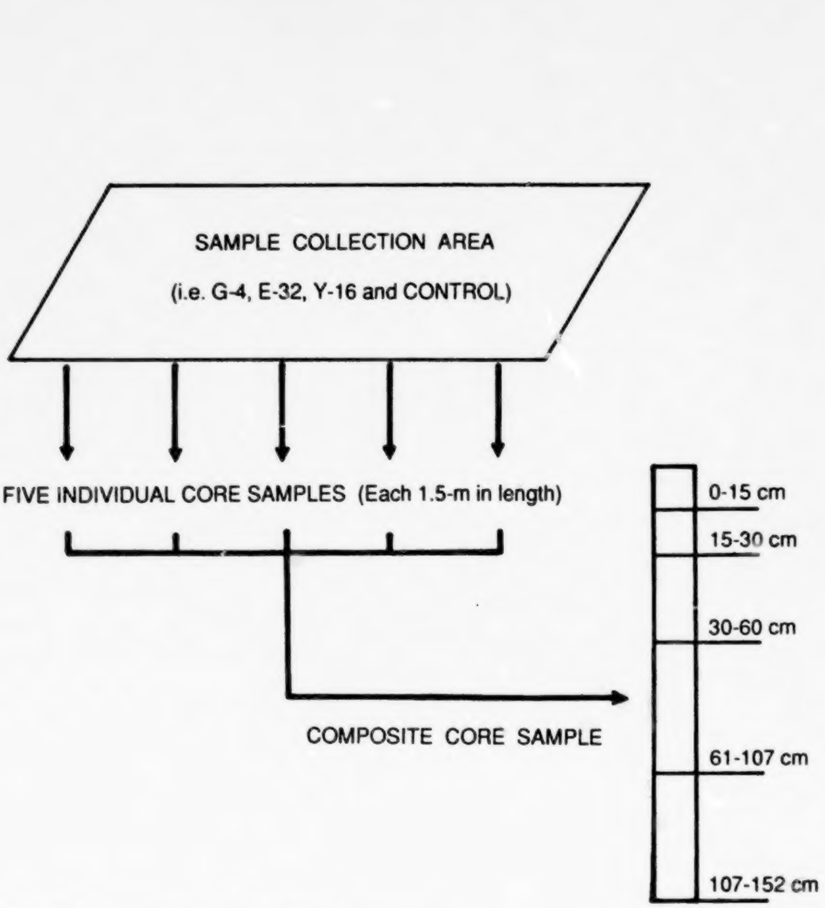

SAMPLE COLLECTION AREA
(i.e. G-4, E-32, Y-16 and CONTROL)
FIVE INDIVIDUAL CORE SAMPLES (Each 1.5-m in length)
COMPOSITE CORE SAMPLE
0-15 cm
15-30 cm
30-60 cm
61-107 cm
107-152 cm

Figure 7-4. Core Sample Compositing Sequence

RESULTS

The composited core samples were subjected to a variety of analyses to aid in the evaluation of the effects of long-term land treatment on soil characteristics. The results of these analyses are presented in this section.

Organic Carbon

Composite soil core samples from the G-4, Y-16, E-32 and control areas (Figure 7-1) were analyzed for organic carbon as described in Section 3. Each sample was analyzed in duplicate. Mean values and standard deviations at five different depth intervals for each sampling area were plotted (Figures 7-5 through 7-8).

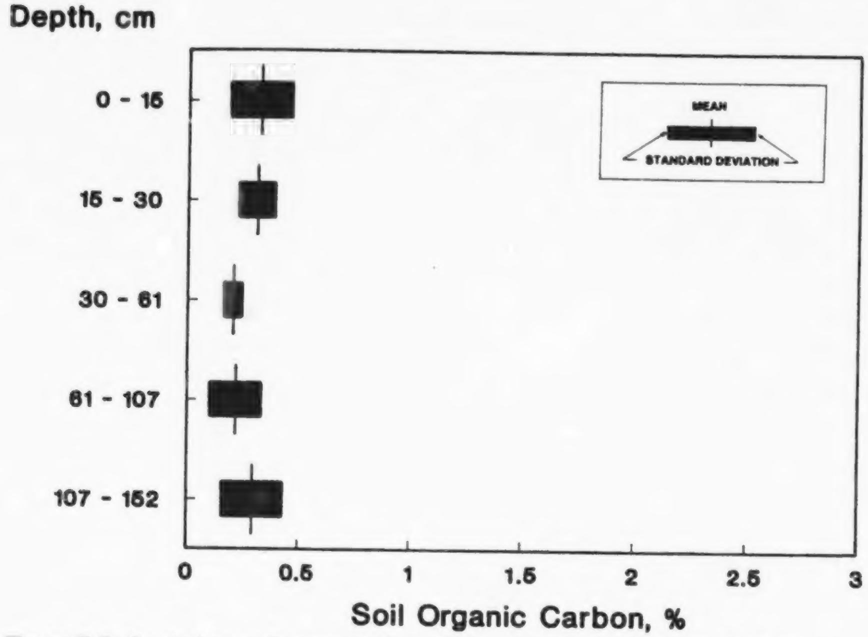


Figure 7-5. Control Area Composite Core Samples: Percent Organic Carbon

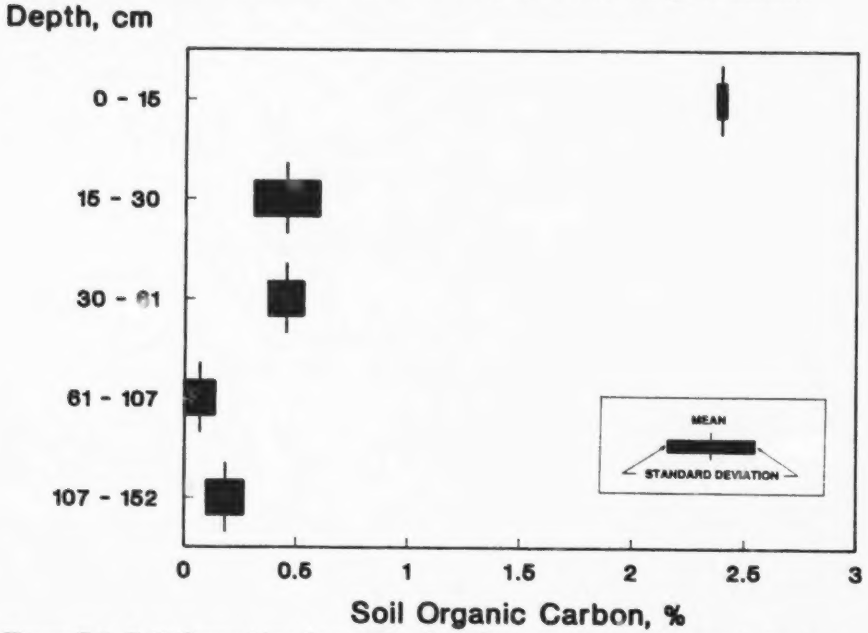


Figure 7-6. E-32 Spray Line Composite Core Samples: Percent Organic Carbon

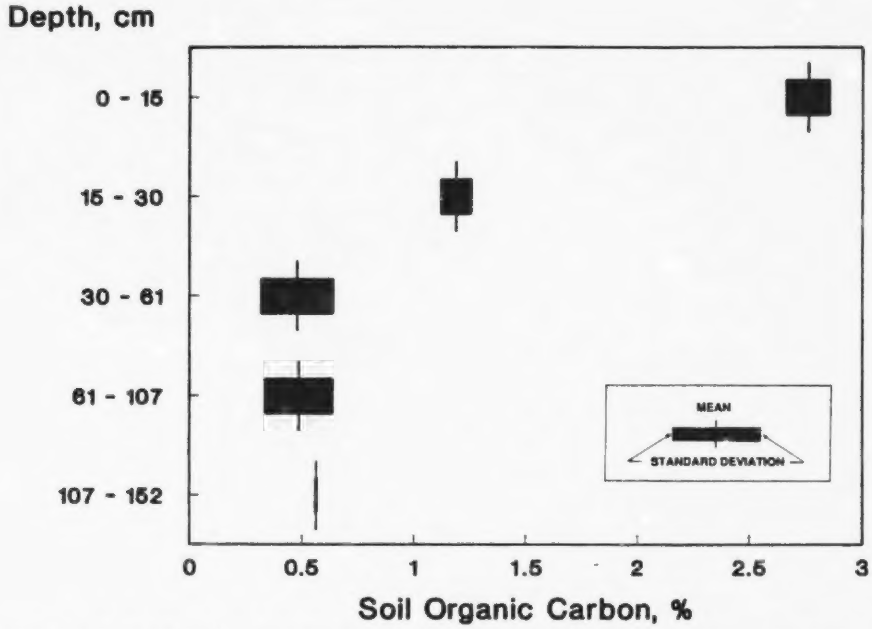


Figure 7-7. Y-16 Spray Line Composite Core Samples: Percent Organic Carbon

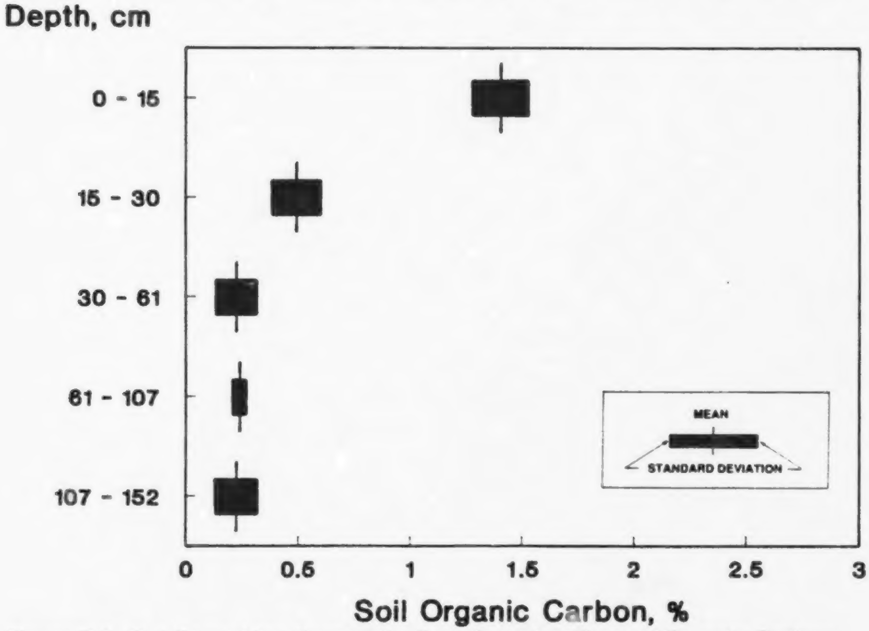


Figure 7-8. G-4 Spray Line Composite Core Samples: Percent Organic Carbon

The organic carbon data were subjected to two way analysis of variance (ANOVA2) to test hypotheses of equality of the means with respect to location and depth. The ANOVA2 results are valid for *all locations* and *all depths*, with a 1 in 20 chance of error, at a 5% level of significance.

For the Paris, TX situation, identical depth intervals were used at all locations; and thus, depth was considered a fixed variable. Because the sample collection locations were selected randomly from the hundreds of spray lines at the site, location was considered a random variable. Thus, it was appropriate to test the significance of depth and location even when interaction effects were significant.

The organic carbon data were arranged in a matrix, grouped by location (the three spray lines and the control area) and depth (five depth intervals). Each location/depth interval contained three replicate analyses. Where significant differences (at the 5% level of significance) between the means were found to exist, the possible differences were examined and identified. The Q-method(8) was used to examine the possibility of differences between the means (as a function of depth, location or both). The results of the ANOVA2 and Q-method testing are presented in Table 7-1.

TABLE 7-1. Q-METHOD TEST RESULTS FOR ORGANIC CARBON: LOCATION

Location	**Control**	**G-4 Line**	**E-32 Line**	**Y-16 Line**
Mean, %	0.27 **a**	0.52 **b**	0.71 **c**	1.00 **d**

Depth, cm	**0 to 15**	**15 to 30**	**30 to 61**	**61 to 107**	**107 to 152**
Mean, %	1.72 **a**	0.61 **b**	0.34 **c**	0.26 **c**	0.32 **c**

At a given location/depth, concentration values followed by the same letter are not significantly different at the 5% level.

Location was significant at 5% level, ${}^{3}_{20}F = 58.6$, critical ${}^{3}_{20}F = 3.10$.

Depth was significant at 5% level, ${}^{4}_{20}F = 146$, critical ${}^{4}_{20}F = 2.87$.

Interaction was significant at 5% level, ${}^{12}_{20}F = 19.79$, critical ${}^{12}_{20}F = 2.12$.

Recalling that at the time of sample collection the E-32 line had been in operation for 10-years and the Y-16 and G-4 lines for 23-years data from these areas can be used to assess the effect of the length of time (number of years) of application on soil properties. The ANOVA2 and Q-method test results indicated that the organic carbon percentage was significantly different at each location, with no two locations being statistically similar. If it is assumed that the initial soil organic carbon concentration at all three wastewater application areas was roughly equivalent, and that the operation of the land treatment system resulted in the addition of organic matter to the soil surface, it followed that the oldest application areas would possess a greater amount of soil organic carbon than the more recently developed areas. Although the soil in wastewater application areas showed a definite increase in organic carbon content as a result of wastewater application, the distribution of mean organic carbon content did not follow the expected trend. The reason for this may be due to differences in harvesting practices at the locations over the years, variations in the application rates at the different lines, differences in the initial soil organic carbon percentage prior to wastewater application, or differences in soil texture or properties which serve to retain the organic carbon on the soil surface.

The organic carbon percentage followed the trend of high values at the surface (0 to 15-cm and 15 to 30-cm) which decreased to a statistically uniform concentration after 30-cm of depth. Assuming that that original organic carbon percentage at each location was near or equal to the value measured at the control area these results indicated that deposition of organic matter at the site was variable with no apparent correlation between organic carbon accumulation and time. Surface accumulations of organic carbon at the waste application areas were significantly greater than that measured at the control area.

pH and Cation Exchange Capacity

Cation Exchange Capacity (CEC) and pH values for the composite soil core samples were measured as described in Chapter 3. CEC is normally expressed as the number of milliequivalents of cations that can be exchanged in a sample with a dry mass of 100-g(6).

The measured pH and CEC values are tabulated in Table 7-2.

TABLE 7-2. pH AND CEC VALUES: COMPOSITE SOIL CORES

Location	**Control**	**G-4**	**Y-16**	**E-32**
Depth, cm	**pH / CEC***	**pH / CEC***	**pH / CEC***	**pH / CEC***
0-15	6.56 / 32	5.84 / 17	5.67 / 51	5.65 / 59
15-30	4.86 / 37	5.04 / 25	5.85 / 53	5.11 / 65
30-61	4.65 / 42	4.87 / 55	5.50 / 55	6.10 / 57
61-107	5.27 / 48	5.08 / 51	6.56 / 66	6.64 / 49
107-152	5.55 / 45	6.06 / 20	6.80 / 61	7.16 / 56

* CEC in units of meq/100-g of dry soil.

The CEC data were tested by ANOVA2 and no significant difference ($p \leq 0.05$) between the means was found with respect to depth. However, there was a significant difference with respect to location. The location data were tested using the Q-method(8) (Table 7-3).

TABLE 7-3. Q-METHOD TEST RESULTS FOR CEC: LOCATION

Location	**Control**	**G-4 Line**	**E-32 Line**	**Y-16 Line**
Mean, mg/kg	40.8 **a**	33.6 **a**	57.2 **b**	57.2 **b**

At a given location, concentration values followed by the same letter are not significantly different at the 5% level.
Location was significant at 5% level, ${}^{3}_{12}F = 6.96$, critical ${}^{3}_{12}F = 3.49$.

The CEC values were statistically uniform with depth at each location; however, the mean values at lines E-32 and Y-16 were statistically different from those at the control area and the G-4 line. The CEC value represents the ultimate ability of the soil clay mineral fraction to replace cation adsorbed on the surface with cations in the surrounding soil solution. Because the CEC test is a measure of the ultimate, theoretical exchange ability of the clay and not the in-situ exchange capacity remaining after equilibration with soil pore water, no valid comparisons can be made between CEC and length of time of site usage.

The principal utility of CEC data is as an aid in the evaluation of the potential for cation accumulation and/or migration through soil. The application of CEC data to this end is described later in this Chapter.

Generally, it is unacceptable to calculate mean pH values based on the hydrogen ion concentration. Averaging the non-conservative quantity $[H^+]$ yields erroneously low mean pH values.(14) Because of this, statistical evaluation of the pH data was not performed. On a qualitative basis, however, inspection of the pH data indicated that the soil pH range extended from acidic to neutral (4.65 to 7.16), with the distribution skewed to the acidic condition. An examination of the relation between soil solution pH and soil metal retardation capacity is presented later in this Chapter.

Total Metals

The composite core samples were analyzed in triplicate for the total metals noted in Table 7-4. Based on the characteristics of the applied wastewater and typical soils, it was anticipated that these 24 metals would be present in trace or greater concentrations in the soil. Of the 24 metals analyzed for, the data from zinc, chromium, nickel, potassium, magnesium, sodium, and calcium were subjected to further examination and statistical analysis. These seven metals were selected because of their presence and relative magnitude of concentrations in both the raw wastewater and soil. Samples from each location and depth were analyzed in triplicate, mean concentrations and standard deviation values are provided in Table 7-5.

TABLE 7-4. METALS EXAMINED IN SOIL CORE SAMPLES

Magnesium	Sodium	Beryllium
Iron	Potassium	Copper
Manganese	Cobalt	Strontium
Lead	Lithium	Aluminum
Nickel	Molybdenum	Vanadium
Arsenic	Barium	Calcium
Selenium	Boron	Chromium
Cadmium	Titanium	Zinc

TABLE 7-5. SOIL METAL CONCENTRATIONS: MEANS AND STANDARD DEVIATIONS

Depth Interval	Total calcium concentration in soil, mg/kg dry weight[*]			
	Control	G-4 line	E-32 line	Y-16 line
0 to 15-cm	4,220 ± 489	1,357 ± 97	6,390 ± 1,161	4,280 ± 2,818
15 to 30-cm	3,027 ± 477	1,680 ± 291	6,380 ± 347	4,625 ± 601
30 to 61-cm	10,800[**]	2,967 ± 266	6,483 ± 765	4,970 ± 252
61 to 107-cm	30,367 ± 9,717	5,557 ± 1,521	16,467 ± 1,955	8,333 ± 843
107 to 152-cm	32,333 ± 18,939	4,825 ± 530	12,533 ± 666	6,960 ± 3,168

Depth Interval	Total chromium concentration in soil, mg/kg dry weight[*]			
	Control	G-4 line	E-32 line	Y-16 line
0 to 15-cm	21 ± 9	20 ± 5	21 ± 2	26 ± 2
15 to 30-cm	25 ± 2	28 ± 7	24 ± 7	30 ± 3
30 to 61-cm	17[**]	29 ± 2	23 ± 7	21 ± 10
61 to 107-cm	22 ± 13	29 ± 6	17 ± 3	21 ± 5
107 to 152-cm	17 ± 3	38 ± 10	19 ± 1	29 ± 11

Depth Interval	Total magnesium concentration in soil, mg/kg dry weight[*]			
	Control	G-4 line	E-32 line	Y-16 line
0 to 15-cm	1,071 ± 257	423 ± 34	2,440 ± 337	2,430 ± 1,050
15 to 30-cm	1,597 ± 192	1,350 ± 316	3,273 ± 186	2,388 ± 325
30 to 61-cm	1,760+[*]	2,126 ± 120	3,250 ± 690	1,473 ± 403
61 to 107-cm	2,067 ± 290	2,487 ± 380	2,913 ± 301	2,047 ± 404
107 to 152-cm	2,327 ± 283	3,470 ± 721	3,160 ± 283	2,235 ± 955

Notes are explained at the end of the table.

TABLE 7-5. (Continued)

Depth Interval	Total nickel concentration in soil, mg/kg dry weight*			
	Control	G-4 line	E-32 line	Y-16 line
0 to 15-cm	11 ± 3	8 ± 2	25 ± 5	18 ± 7
15 to 30-cm	11 ± 2	10 ± 4	30 ± 2	16 ± 2
30 to 61-cm	25**	14 ± 5	36 ± 12	21 ± 3
61 to 107-cm	12 ± 5	19 ± 2	24 ± 2	17 ± 5
107 to 152-cm	9 ± 2	33 ± 6	22 ± 1	22 ± 0

Depth Interval	Total potassium concentration in soil, mg/kg dry weight*			
	Control	G-4 line	E-32 line	Y-16 line
0 to 15-cm	608 ± 44	454 ± 23	1,687 ± 336	1,650 ± 660
15 to 30-cm	768 ± 167	1,111 ± 611	2,243 ± 609	1,085 ± 78
30 to 61-cm	558**	987 ± 186	2,067 ± 887	588 ± 43
61 to 107-cm	719 ± 173	1,082 ± 267	1,830 ± 251	875 ± 255
107 to 152-cm	920 ± 52	2,000 ± 863	1,970 ± 697	1,066 ± 246

Depth Interval	Total sodium concentration in soil, mg/kg dry weight*			
	Control	G-4 line	E-32 line	Y-16 line
0 to 15-cm	267 ± 122	138 ± 53	518 ± 147	670 ± 384
15 to 30-cm	479 ± 142	456 ± 150	578 ± 447	526 ± 139
30 to 61-cm	1,140**	1,122 ± 223	1,184 ± 258	795 ± 127
61 to 107-cm	2,573 ± 612	1,417 ± 196	1,277 ± 334	991 ± 205
107 to 152-cm	3,157 ± 759	1,645 ± 770	1,314 ± 424	782 ± 171

Notes are explained at the end of the table.

TABLE 7-5. (Continued)

Depth Interval	Total zinc concentration in soil, mg/kg dry weight*			
	Control	G-4 line	E-32 line	Y-16 line
0 to 15-cm	38 ± 14	57 ± 4	67 ± 22	74 ± 21
15 to 30-cm	29 ± 5	29 ± 7	56 ± 4	35 ± 0
30 to 61-cm	25**	41 ± 9	62 ± 14	41 ± 16
61 to 107-cm	31 ± 8	38 ± 7	77 ± 24	35 ± 5
107 to 152-cm	29 ± 3	67 ± 4	63 ± 3	58 ± 5

* Each composite soil sample analyzed in triplicate.

** Replicate samples (extracts) lost, single sample analyzed from *control* area at this depth

The soil concentration data for these metals were subjected to ANOVA2 testing. The testing indicated that, for all seven metals, with the exceptions of potassium and chromium over depth, there was a statistically significant difference between the means with respect to both depth and location. The data were further analyzed by the Q-method[8] and the results for each metal are given in Table 7-6.

TABLE 7-6. Q-METHOD TEST RESULTS FOR POTASSIUM

Location	Control	G-4 Line	E-32 Line	Y-16 Line
Mean, mg/kg	715 a	1,127 b	1,126 b	1,489 b

Depth, cm	0 to 15	15 to 30	30 to 61	61 to 107	107 to 152
Mean, mg/kg	1,100 a	1,302 a	1,050 a	1,126 a	1,489 a

At a given location/depth concentration values followed by the same letter are not significantly different at the 5% level.

Location was significant at 5% level, ${}^{3}_{40}F = 25.7$, critical ${}^{3}_{40}F = 2.84$.

Depth was not significant at 5% level, ${}^{4}_{40}F = 2.41$, critical ${}^{4}_{40}F = 2.61$.

Interaction was significant at 5% level, ${}^{12}_{40}F = 2.42$, critical ${}^{12}_{40}F = 2.00$.

The results indicated that potassium soil concentrations were similar in all three waste application areas; however, the mean concentrations were different (higher) than those found at the control area. The soil potassium levels in the application areas indicated that wastewater application resulted in an increase in potassium in the soil. However, the increase did not appear to be dependent upon the cumulative length of time the spray lines were operational. Potassium concentrations were uniform with depth.

TABLE 7-7. Q-METHOD TEST RESULTS FOR CALCIUM

Location	Control	G-4 Line	E-32 Line	Y-16 Line
Mean, mg/kg	16,149 a	3,277 b	9,651 c d	5,834 b d

Depth, cm	0 to 15	15 to 30	30 to 61	61 to 107	107 to 152
Mean, mg/kg	4,062 a	3,928 a	6,305 a	15,181 b	14,163 b

At a given location/depth, concentration values followed by the same letter are not significantly different at the 5% level.

Location was significant at 5% level, ${}^{3}_{40}F = 19.7$, critical ${}^{3}_{40}F = 2.84$.

Depth was significant at 5% level, ${}^{4}_{40}F = 15.3$, critical ${}^{4}_{40}F = 2.61$.

Interaction was significant at 5% level, ${}^{12}_{40}F = 4.53$, critical ${}^{12}_{40}F = 2.00$.

The calcium data (Table 7-7) indicated that calcium concentrations were lower at the application areas than the control area. With respect to time, the mean soil calcium concentration values were lower at the older areas than at the younger area (and the control). Calcium concentration increased with depth at all locations. These data indicated that leaching of calcium from the top 1.5-m of soil occurred at the wastewater application areas, and the leaching was apparently time dependent. That is, 23-years of wastewater application resulted in more calcium leached than the amount leached in 10-years. A possible mechanism for the apparent leaching is discussed in Chapter 9.

Mean soil concentration values of calcium at the 4 study locations were plotted (Figure 7-9) to illustrate the leaching trend with respect to the unleached soil at the control area.

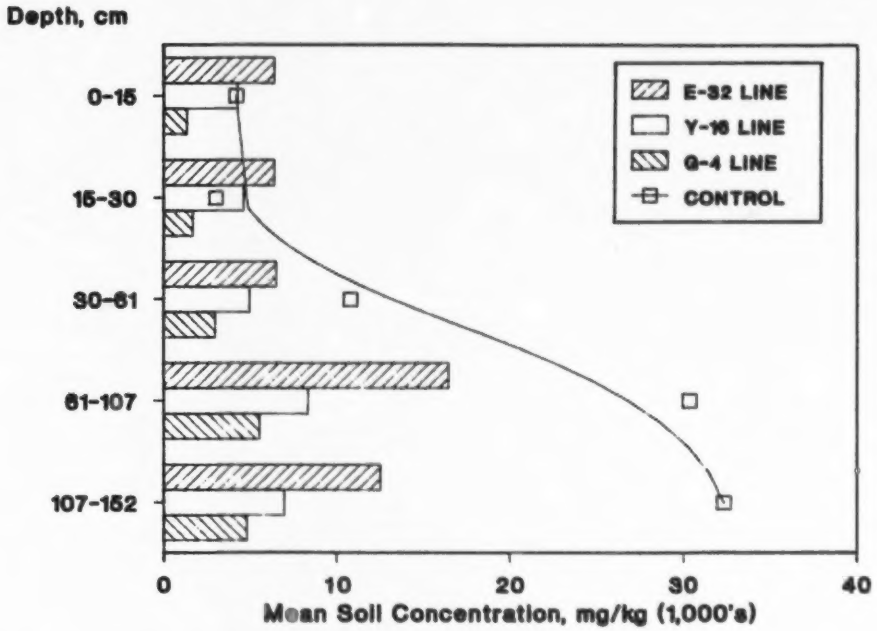


Figure 7-9. Leaching of Calcium From Soil in Wastewater Application Areas

TABLE 7-8. Q-METHOD TEST RESULTS FOR MAGNESIUM

Location	Control	G-4 Line	E-32 Line	Y-16 Line
Mean, mg/kg	1,764 a	1,971 a	2,981 b	2,113 a

Depth, cm	0 to 15	15 to 30	30 to 61	61 to 107	107 to 152
Mean, mg/kg	1,591 a	2,150 b	2,153 b	2,378 b c	2,765 c

At a given location/depth, concentration values followed by the same letter are not significantly different at the 5% level.

Location was significant at 5% level, ${}^{3}_{40}F = 25.2$, critical ${}^{3}_{40}F = 2.84$.

Depth was significant at 5% level, ${}^{4}_{40}F = 12.8$, critical ${}^{4}_{40}F = 2.61$.

Interaction was significant at 5% level, ${}^{12}_{40}F = 6.44$, critical ${}^{12}_{40}F = 2.00$.

Mean magnesium values did not vary significantly by location, with the exception of the higher values at the E-32 line. Mean magnesium concentration values increased with depth at all locations.

TABLE 7-9. Q-METHOD TEST RESULTS FOR SODIUM

Location	**Control**	**G-4 Line**	**E-32 Line**	**Y-16 Line**
Mean, mg/kg	1,523 **a**	956 **b**	1,026 **b**	789 **b**

Depth, cm	**0 to 15**	**15 to 30**	**30 to 61**	**61 to 107**	**107 to 152**
Mean, mg/kg	399 **a**	588 **a**	1,060 **b**	1,539 **c**	1,783 **c**

At a given location/depth, concentration values followed by the same letter are not significantly different at the 5% level.

Location was significant at 5% level, ${}^{3}_{40}F = 14.0$, critical ${}^{3}_{40}F = 2.84$.

Depth was significant at 5% level, ${}^{4}_{40}F = 39.7$, critical ${}^{4}_{40}F = 2.61$.

Interaction was significant at 5% level, ${}^{12}_{40}F = 7.42$, critical ${}^{12}_{40}F = 2.00$.

Mean sodium concentrations were similar in all three waste application areas, but less than that at the control area. The sodium concentrations increased gradually with depth at all four locations. Although the sodium values at the wastewater application areas were less than those at the control area, there was no statistical difference between wastewater application areas, regardless of cumulative time of usage. These data indicate that leaching of sodium from the top 1.5-m of soil occurred in all three wastewater application areas. Mean values of soil sodium concentration have been plotted (Figure 7-10) to illustrate the leaching trend.

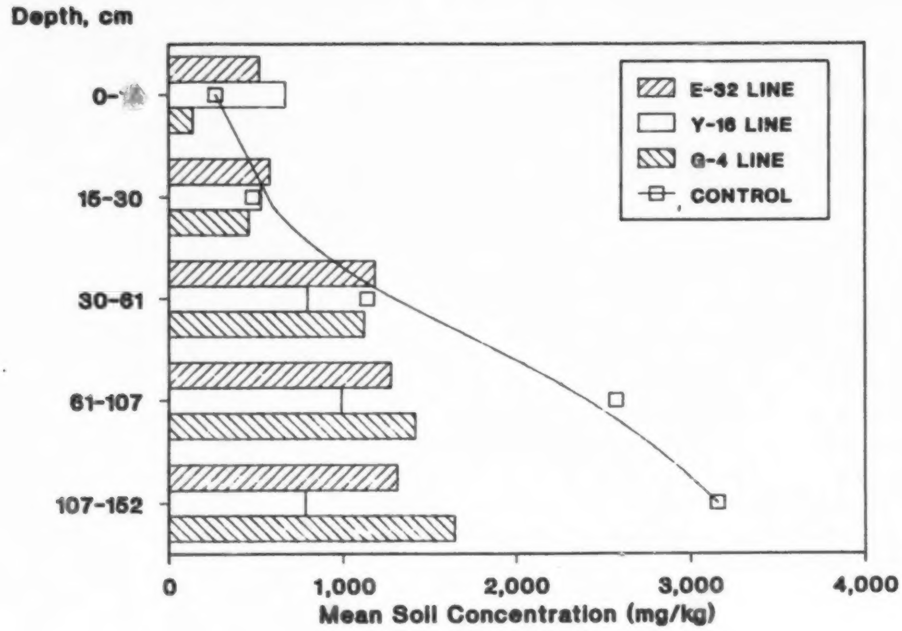


Figure 7-10. Leaching of Sodium From Soil in Wastewater Application Areas

TABLE 7-10. Q-METHOD TEST RESULTS FOR ZINC

Location	Control	G-4 Line	E-32 Line	Y-16 Line
Mean, mg/kg	30 a	47 b	65 c	48 b d

Depth, cm	0 to 15	15 to 30	30 to 61	61 to 107	107 to 152
Mean, mg/kg	59 a c	37 b	43 b	45 b c	54 c

At a given location/depth, concentration values followed by the same letter are not significantly different at the 5% level.

Location was significant at 5% level, ${}^{3}_{40}F = 24.5$, critical ${}^{3}_{40}F = 2.84$.

Depth was significant at 5% level, ${}^{4}_{40}F = 7.61$, critical ${}^{4}_{40}F = 2.61$.

Interaction was significant at 5% level, ${}^{12}_{40}F = 2.42$, critical ${}^{12}_{40}F = 2.00$.

The geatest mean zinc values were found at the youngest application area, contrary to the predicted trend, but similar to that observed for magnesium. Mean zinc concentrations at all wastewater application areas, however, were greater than those at the control area. These data indicate that wastewater application resulted in accumulation of zinc throughout the soil profile. The data also indicate that, for the 5 depths examined, zinc concentrations were greatest at the surface and the lowest (107 to 152-cm) depth.

TABLE 7-11. Q-METHOD TEST RESULTS FOR NICKEL

Location	Control	G-4 Line	E-32 Line	Y-16 Line
Mean, mg/kg	14 a	17 a b	27 c	19 b

Depth, cm	0 to 15	15 to 30	30 to 61	61 to 107	107 to 152
Mean, mg/kg	16 a	17 a	24 c	18 a b	21 b c

At a given location/depth, concentration values followed by the same letter are not significantly different at the 5% level.

Location was significant at 5% level, ${}^{3}_{40}F = 30.7$, critical ${}^{3}_{40}F = 2.84$.

Depth was significant at 5% level, ${}^{4}_{40}F = 8.66$, critical ${}^{4}_{40}F = 2.61$.

Interaction was significant at 5% level, ${}^{12}_{40}F = 7.52$, critical ${}^{12}_{40}F = 2.00$.

Nickel concentrations at the Y-16 and E-32 areas were greater than those at the control area. The greatest nickel concentrations were at the E-32 line contrary to the predicted time-dependent trend, but, similar to that observed for zinc and magnesium.

TABLE 7-12. Q-METHOD TEST RESULTS FOR CHROMIUM

Location	Control	G-4 Line	E-32 Line	Y-16 Line
Mean, mg/kg	21 a	29 b	21 a	25 a b

Depth, cm	0 to 15	15 to 30	30 to 61	61 to 107	107 to 152
Mean, mg/kg	22 a	27 a	23 a	22 a	26 a

At a given location/depth, concentration values followed by the same letter are not significantly different at the 5% level.

Location was significant at 5% level, ${}^{3}_{40}F = 6.95$, critical ${}^{3}_{40}F = 2.84$.

Depth was not significant at 5% level, ${}^{4}_{40}F = 1.79$, critical ${}^{4}_{40}F = 2.61$.

Interaction was not significant at 5% level, ${}^{12}_{40}F = 1.64$, critical ${}^{12}_{40}F = 2.00$.

Chromium concentrations did not vary with depth. Mean chromium values were greatest at the G-4 and Y-16 lines; however, only the chromium concentrations at the G-4 line were statistically different from the control.

Water Soluble Anions

Water soluble extract anion concentrations of chloride and sulfate were measured. The laboratory-measured ion concentrations were less than those existing in the soil due to the dilution of the concentration by the extracting solution. In order to estimate the concentration of each chemical species existing in the soil, c_i, at the time of sample collection, the measured concentration, c_{mi}, was multiplied by the inverse of the dilution factor as follows:

$$c_i = c_{mi}\left(\frac{W_{sol}}{W_w}\right) \qquad (7-1)$$

W_{sol} is the weight of the extracting solution and W_w is the weight of the water in the soil at the time of collection. The concentration, c_i, represents the total concentration of the chemical species in the soil assuming the extraction solution is 100 percent efficient and complexation and precipitation of the dissolved species does not occur[15].

Pore water concentrations were calculated using the water soluble extract data collected for sulfate and chloride. The calculated values are presented below (Table 7-13).

TABLE 7-13. CHLORIDE AND SULFATE PORE WATER CONCENTRATIONS*

Depth, cm	Control	G-4	E-32	Y-16
0 - 15	65 (10,962)	590 (2,660)	115 (911)	295 (1,526)
15 - 30	65 (8,381)	275 (2,543)	166 (775)	300 (832)
30 - 61	300 (24,242)	278 (1,494)	330 (11,325)	588 (1,393)
61 - 107	1,059 (26,428)	1,481 (11,778)	398 (13,248)	604 (8,935)
107 - 152	1,531 (26,059)	704 (3,500)	426 (10,124)	700 (6,395)

* Concentration in units of mg/L. Chloride values followed by sulfate values in parenthesis.

The data for each ion were subjected to ANOVA2 analysis with respect to depth and location. The Q-test results are given in Tables 7-14 and 7-15.

TABLE 7-14. Q-METHOD TEST RESULTS FOR SULFATE

Location	**Control**	**G-4 Line**	**E-32 Line**	**Y-16 Line**
Mean, mg/L	19,214 **a**	4,395 **b**	7,277 **b**	3,817 **b**

Depth, cm	**0 to 15**	**15 to 30**	**30 to 61**	**61 to 107**	**107 to 152**
Mean, mg/L	4,015 **a**	3,133 **a**	9,615 **b**	15,097 **b**	11,519 **b**

At a given location/depth, concentration values followed by the same letter are not significantly different at the 5% level.

Location was significant at 5% level, ${}^{3}_{12}F = 18.8$, critical ${}^{3}_{12}F = 3.49$.

Depth was significant at 5% level, ${}^{4}_{12}F = 7.47$, critical ${}^{4}_{12}F = 3.26$.

TABLE 7-15. Q-METHOD TEST RESULTS FOR CHLORIDE

Location	Control	G-4 Line	E-32 Line	Y-16 Line
Mean, mg/L	604 a	666 a	287 a	497 a

Depth, cm	0 to 15	15 to 30	30 to 61	61 to 107	107 to 152
Mean, mg/L	266 a	202 a	374 a	885 b	840 b

At a given location/depth, concentration values followed by the same letter are not significantly different at the 5% level.

Location was not significant at 5% level, ${}^{3}_{12}F = 1.34$, critical ${}^{3}_{12}F = 3.49$.

Depth was significant at 5% level, ${}^{4}_{12}F = 4.09$, critical ${}^{4}_{12}F = 3.26$.

Calculated mean sulfate pore water concentrations at the three waste application areas were statistically similar; however, they were substantially less than the mean value at the control area. Thus, although leaching appears to have occured, the process was apparently not a direct function of cumulative time of leaching. The results indicate that leaching of sulfate from the top 1.5-m of soil in the wastewater application areas has occurred. The leaching trend is illustrated in Figure 7-11, mean values of sulfate were plotted. Mean sulfate concentrations increased gradually with depth at all locations.

The ANOVA2 and Q-test indicated that mean calculated pore water chloride concentrations, with respect to location, were not statistically different. However, chloride concentrations exhibited the trend of increasing with depth at all locations.

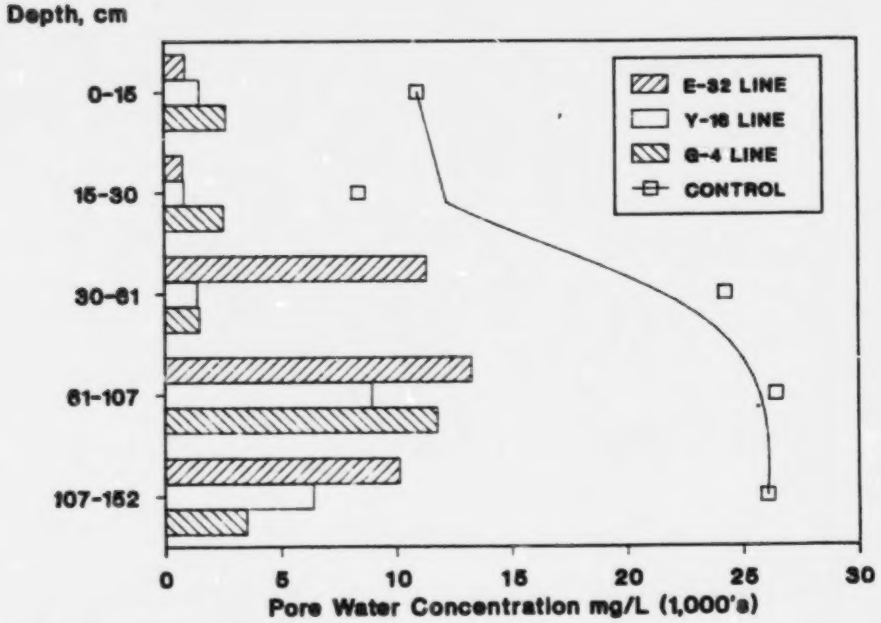


Figure 7-11. Leaching of Sulfate From Soil in Wastewater Application Areas

SUMMARY

The results of the statistical analyses presented in the preceeding section have been summarized in Table 7-15 to highlight the major points.

TABLE 7-16. STATISTICAL SUMMARY OF SOIL CHARACTERISTICS

Soil Component	Was Accumulation Significant?	Was Accumulation Time Dependent?	Was Leaching Significant?	Was Leaching Time Dependent?
% OC	YES, all lines	NO	NO	N/A
Potassium	YES, all lines	NO	NO	N/A
Calcium	NO	N/A	YES, all lines	YES
Magnesium	YES, E-32 only	NO	NO	N/A
Sodium	NO	N/A	YES, all lines	NO
Zinc	YES, all lines	NO	POSSIBLY	N/A
Nickel	YES, all lines	NO	POSSIBLY	N/A
Chromium	NO	N/A	NO	N/A
Sulfate	NO	N/A	YES, all lines	NO
Chloride	NO	N/A	POSSIBLY	NO

N/A Not applicable.

SOIL METAL RETENTION CORRELATIONS

Mean values from all locations and all depths for soil properties and ion concentrations at the wastewater application areas and the control area were tested for correlation through linear regression analysis. A total of 15 data points per variable (i.e., CEC, pH, organic carbon fraction, and soil zinc, nickel and chromium concentration) were used in the correlation analysis. The correlation analysis was performed to aid in the assessment of the ability of soil at the OLF site to immobilize cations.

The correlation coefficient, r, was calculated and ANOVA was performed for the linear regression of Y on X, where Y represents the dependent variable (soil metal concentration) and X represents the independent variable (either %OC, pH, or CEC). The independent variables were selected for the regression analysis because they are often used(16)(17) as an indicator of the ability of a soil to immobilize metals. The one-way ANOVA test of the regression analysis provided a quantitative means of assessing the significance of the regression. F-statistic values were calculated and evaluated at the 5% level of significance. Correlation coefficient values, r, were calculated to determine the direction of the slope of the regression line; i.e., positive r (slope) values indicated that X and Y increased together, negative values indicated that increases in X resulted in decreases in Y.

USEPA guidance documents(16)(17) recommend soil pH and CEC as relative measures of the potential for the retention of *metals of major concern* (Pb, Zn, Cu, Ni, Cd) on agricultural land. The guidelines are rough estimates and are not suitable for all metals. For example, EPA recommends that soil pH be maintained at 6.5 or greater to minimize metal solubilities. Soil pH, however, may vary significantly with depth at a given location, making uniform predictions impossible. This situation was present at the Paris, TX site (Table 7-2), where pH was found to vary between acidic (pH=5) to neutral (pH=7). Also, pH can change dramatically due to acidification resulting from increased nitrification and stimulation of naturally occurring organic acids. Based on the characteristics of the applied wastewater and the expected chemical and biological transforamtions which are expected to take place at an OLF site, increased acidification may occur at the soil surface and at greater depths. In addition, the solubility of many metal complexes can increase at pH values greater than 6.5.

Of the 7 metals studied in detail, 3 metals (zinc, chromium, and nickel) were selected for the regression analyses. These heavy metals were selected because their presence is often a concern at land treatment sites and their retention by soil has been reported(16)(17) to be affected by the independent variables described above. The results of the linear regression and ANOVA testing are given in Table 7-17.

TABLE 7-17. SOIL METAL CORRELATION: LINEAR REGRESSION* AND ANOVA RESULTS

Dependent Variable■	Independent Variable	Correlation Coefficient, r	Regression F-statistic++	Was Regression Significant?
Zinc	pH•	- 0.456	3.42	NO
	%OC	+ 0.304	1.33	NO
	CEC••	- 0.023	0.01	NO
Nickel	pH•	- 0.167	0.37	NO
	%OC	- 0.196	0.52	NO
	CEC••	+ 0.298	1.27	NO
Chromium	pH•	+ 0.048	0.03	NO
	%OC	- 0.085	0.10	NO
	CEC••	- 0.402	2.50	NO

* Data from all three wastewater application areas and all depths were used in the regression analysis.

■ Soil metal concentration value in units of mg/kg.

++ Critical $^{1}_{13}F = 4.67$.

• Hydrogen ion concentration, mole/L used, i.e., antilog of negative pH value.

•• CEC in units of meq per 100-g dry soil.

Several authors(18)(19) have shown CEC and pH as unlikely to be reliable guides for predicting heavy metal mobility in heterogeneous soils. The data in Table 7-17 indicate that for chromium, nickel and zinc, there was no significant linear correlation between the variables listed. In addition, the slopes of the regression lines exhibited no trends and appeared to be random. The lack of correlation may indicate that existing guidelines for metal immobilization are invalid; however, the relatively low soil metal concentrations, small concentration differences between locations (no order of magnitude differences), variability of the data with depth, the metal complex present, as well as effects due to analytical variation all have an affect on the resolution of the linear regression.

Thus, the data appear to support the assertion(18)(19) that pH and CEC (as well as organic carbon content) may not be reliable indicators of the ability of the soil to immobilize metals. These results, however, are valid only for the concentration ranges found at the site.

METAL ACCUMULATION AND ITS EFFECT ON SITE LIFE

It appears that the long-term metal loading rates (Table 5-4) resulted in the accumulation of potassium, magnesium, zinc, and nickel in the soil at the wastewater application areas. The location (site age) specific metals data, however, did not exhibit accumulation trends which would suggest dependence upon the cumulative time of operation, pH, percent organic carbon, or cation exchange capacity. Because of the apparent random pattern of accumulation, metal accumulation predictions were not location specific and could have been applied to the entire site area.

Although accumulation occurred, the resulting concentrations of zinc and nickel were well below action levels recommended by U.S.EPA(16)(17). In general, zinc and nickel will be toxic to crops before their concentration in plant tissues reaches a level that poses a problem to forage animals or human health. Although this guideline is useful, calculation of the expected site life provided a meaningful value. The calculations were based upon estimates of long-term metal (Zn and Ni) loading rate ranges, present wetted acreage, EPA recommended cumulative limits(16)(17), and the most conservative assumption that 100% of the applied metals were retained by the soil. Because potassium and magnesium accumulations do not represent a potential land management concern, these elements were not considered in the calculation of site life.

The assumptions made for the calculation of soil metal accumulation rates are presented in Table 7-18. The reasoning or data sources used to make the assumptions also are provided (see *Comments* column Table 7-18).

TABLE 7-18. ASSUMPTIONS USED FOR THE CALCULATION OF METAL ACCUMULATION

Variable	Assumption	Comments
Operating period	350 days/year	2 week shutdown/year.
Wetted Acreage	364 hectare	1989 site size.
Initial Soil Metal Concentration	Zinc: 65 mg/kg Nickel: 27 mg/kg	Maximum mean values from field data.
Soil Density	2.00 g/cm^3	Estimated value.
Wastewater Application Rate	16,100 m^3/d	Based on long-term data in Table 5-2.
Applied Wastewater Metal Concentration	Zinc: 0.17 mg/L Nickel: 0.01 mg/L	Zinc value from Table 5-2. Nickel value estimated.
Mean Applied Metal Load and Standard Deviation, kg/d	Zn: 3.0 ± 2.0 kg/d Ni: 0.2 ± 0.2 kg/d	Zinc values from Table 5-4. Nickel values estimated.
Depth of Metal Accumulation In the Soil Profile	Top 15-cm	Assuming that most accumulation occurs at the surface.
Percentage Retained	100%	Most conservative case
EPA Recommended(16)(17) Soil Metal Limits	Zinc: 1,120 kg/ha Nickel: 560 kg/ha	Assuming pH > 6.5 and CEC > 15 meq/100-g.

The results of the metal accumulation calculations are presented in Table 7-19. Although metal accumulation occurred and may be expected to continue, at current metal loading rates, the accumulation of zinc and nickel will not affect site life. These results are valid for mean loading estimates plus and minus one standard deviation.

TABLE 7-19. CALCULATION OF METAL ACCUMULATION AND SITE LIFE EXPECTANCY

Results	Zinc	Nickel
Initial Soil Metal Load, 1987	200 kg/ha	85 kg/ha
Corrected Soil Metal Limits*	920 kg/ha	475 kg/ha
Site Life Expectancy:		
Mean Value	320 years	2,470 years
Low Value	1,000 years	indefinite
High Value	190 years	1,240 years

* Corrected limit value = [EPA recommended value] - [initial metal load].

CONCLUSIONS

Several statistically significant trends were discerned for the total metal soil concentration data and the calculated pore water concentrations. With respect to the control area, accumulation of organic carbon, potassium, magnesium, zinc, and nickel in the soil at the wastewater application areas was evident, as well as leaching of calcium, sodium, and sulfate and possibly chloride, zinc, and nickel. Neither accumulation nor leaching of chromium was evident.

Although the accumulation of zinc and nickel was evident, the accumulation was so small that several hundred years of continued site use may be expected at present loading rates before the cumulative zinc and nickel soil concentrations would approach action levels recommended by EPA for agricultural lands.

CHAPTER 8

HYDROGEOLOGY

INTRODUCTION

A comprehensive review of the literature related to the hydrogeology of north-central Texas, which includes Lamar County in which Paris, Texas is located, was made. The geologic information was utilized during the assessment of the potential for groundwater occurrence at the Paris, Texas site.

The data in the literature indicated that the probability of groundwater existing in recoverable quantities at the Campbell's site was very low. A number of deep borings, however, were made, both on- and off-site to assess the actual conditions at the site. Three groundwater monitoring wells were installed and sampled approximately quarterly between 1987 and 1989.

A complete description of regional hydrogeology and the results of the groundwater sampling program are provided in this chapter.

GENERAL GEOLOGY

Geologic History

The present structural attitude and stratigraphic succession of geologic formations in north-central Texas resulted from a a sequence of events in geologic time. A continuing cycle of advance and retreat of ancient seas resulted in periods of sediment accumulation alternating with periods of erosion. Subsequent structural deformation altered the attitude of the strata. Nonmarine deposition of sediment by streams and other water bodies completed the depositional sequence. An index map of major structural features of the East Texas basin is provided in Figure 8-1.

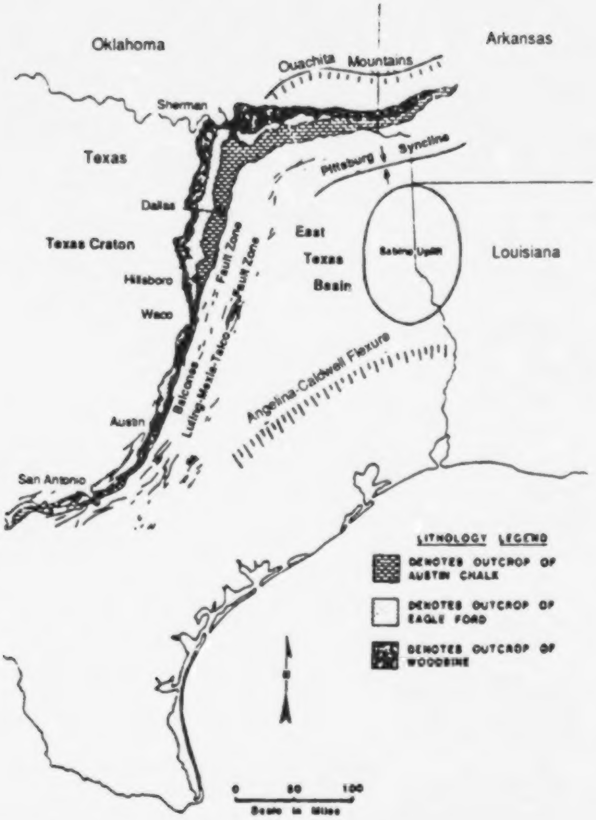

Oklahoma
Arkansas
Ouachita Mountains
Sherman
Texas
Pittsburg Syncline
Dallas
Texas Craton
East
Texas
Basin
Sabine Uplift
Louisiana
Hillsboro
Waco
Fault Zone
Fault Zone
Balcones
Luling-Mexia-Talco
Angelina-Caldwell Flexure
Austin
San Antonio
LITHOLOGY LEGEND
DENOTES OUTCROP OF AUSTIN CHALK
DENOTES OUTCROP OF EAGLE FORD
DENOTES OUTCROP OF WOODBINE
N
0
50
100
Scale in Miles

Figure 8-1. Index Map of Major Structural Features of the East Texas Basin

Paleozoic

During most of the Paleozic era, a sedimentary basin existed throughout much of central and north-central Texas which received sediments that resulted in the formation of sandstone, limestone, and carbonaceous shales. Sediments were deposited in this basin until late Pennsylvanian time when the Llano Uplift and the Ouachita Fold Belt (Figure 8-1) caused a regional tilting to the west and faulting in the immediate uplift area. During Permian time, the basin shifted to the west and only the northwest corner continued to receive sediments. The rest of the basin (inclusive of Lamar County) underwent extensive erosion[20].

Cretaceous

During the Triassic and Jurassic Periods of the early Mesozoic era, withdrawal of the seas from the north-central Texas area along with subsidence in the Gulf Coast embayment led to a reversal of drainage direction. By the close of Jurassic time, Paleozoic rocks had been reduced to an almost flat-featureless plain upon which marine sediments were deposited along an oscillating shoreline during the Cretaceous period.

Two major invasions of the seas during the Cretaceous period are represented by the Comanche and Gulf Series (rocks formed during the Cretacous period). During late Cretaceous (Gulf Series), a general uplift occurred to the west and the seas withdrew gulfward covering only the eastern portion north-central Texas(20).

Tertiary and Quaternary

At the close of the Cretaceous period, noted by uplifting to the west and subsidence of the coastal area, sediments of the Tertiary and Quaternary age were deposited. The repeated transgression and regression of the sea resulted in an alternating sequence of marine and continental deposits. Throughout Tertiary time, except for minor periods of subsidence, the land surface was eroded and modified by streams. During Quaternary time, the streams deposited alluvial sediments. The older sediments are represented by terrace deposits above the alluvial valleys of present streams(20).

Stratigraphy

Stratigraphic units that supply fresh to slightly saline water to wells in north-central Texas range from Paleozoic to Recent. However, the most important water bearing formations are of Cretaceous age. The Stratigraphic units of North-Central Texas are listed in Table 8-1.

TABLE 8-1. STRATIGRAPHIC UNITS AND GEOLOGIC TIME SCALE

Era	System	Series	Group	Stratigraphic units	
Cenozoic	Quaternary	Recent		Alluvium	
		Pleistocene		Fluviatile terrace deposits	
	Tertiary	Eocene	Wilcox		
		Paleocene	Midway		
Mesozoic	Cretaceous	Gulf	Navarro	Kemp Clay Corsicana Marl	
				Nacatoch Sand	
			Taylor	Marlbrook Marl Pecan Gap Chalk Wolfe City - Ozan Formations	
			Austin	Gober Chalk Brownstown Marl Blossom Sand Bonham Formation	
			Eagle Ford		
			Woodbine		
		Comanche	Washita	Grayson Marl - Mainstreet Limestone Pawpaw Formation - Weno Limestone - Denton Clay Fort Worth - Duck Creek Kiamichi Formation	
			Fredericksburg	Edwards Limestone Comanche Peak Formation	Goodland Limestone
				Walnut Formation	
			Trinity	Antlers Formation	Paluxy Formation
					Glen Rose Formation
					Twin Mountains Formation
Paleozoic				Paleozoic rocks undifferentiated	

The Cretaceous System is composed of two series, Gulf and Comanche, and each is divided into groups. The Gulf Series is divided into the following five groups: Navarro, Taylor, Austin, Eagle Ford, and Woodbine. The Comanche Series is divided into the following three groups: Washita, Fredericksburg, and Trinity.

The Taylor, Austin, and Eagle Ford groups outcrop in Lamar County (Figure 8-2, Geologic Outcrop Map: North-Central Texas). The Austin and Eagle Ford groups were of particular interest to this study because groundwater monitoring wells at the Paris, Texas site were completed within them. Stratigraphic units within the Austin formation which outcrop at the Paris, Texas site include the Blossom Sand and the Bonham Formation.

Eagle Ford rocks unconformably overlie (evidenced by the break in the normal depositional sequence) the Woodbine strata along the margins of the East Texas basin. The Eagle Ford Group is overlain by the Austin Group in all portions of the basin. Throughout the basin, this contact is believed to be unconformable(21).

Lithology and Groundwater Potential

Blossom Sand Formation - Austin Group

The Blossom Sand of the Austin Group crops out in central Fannin, Lamar, and Red River Counties in north-central Texas. The strike (bedding plane as it intersects the horizontal) of the outcrop is east-west. The dip (the angle that a planar feature makes with the horizontal) of the Blossom Sand is to the south averaging about 16.1 m/km.

The Blossom Sand consists of a brown to light gray, unconsolidated, calcareous, ferruginous, glauconite fine to medium quartz sand interbedded with sandy clay and chalky marl. Most of the formation contains impermeable sandy clay or marl and chalk, with only about 25 percent sand.

The Blossom thickens southward downdip and eastward along the strike. Thickness varies from 6 to 76-m thinning westward. Individual beds, for the most part extend over several square kilometers.

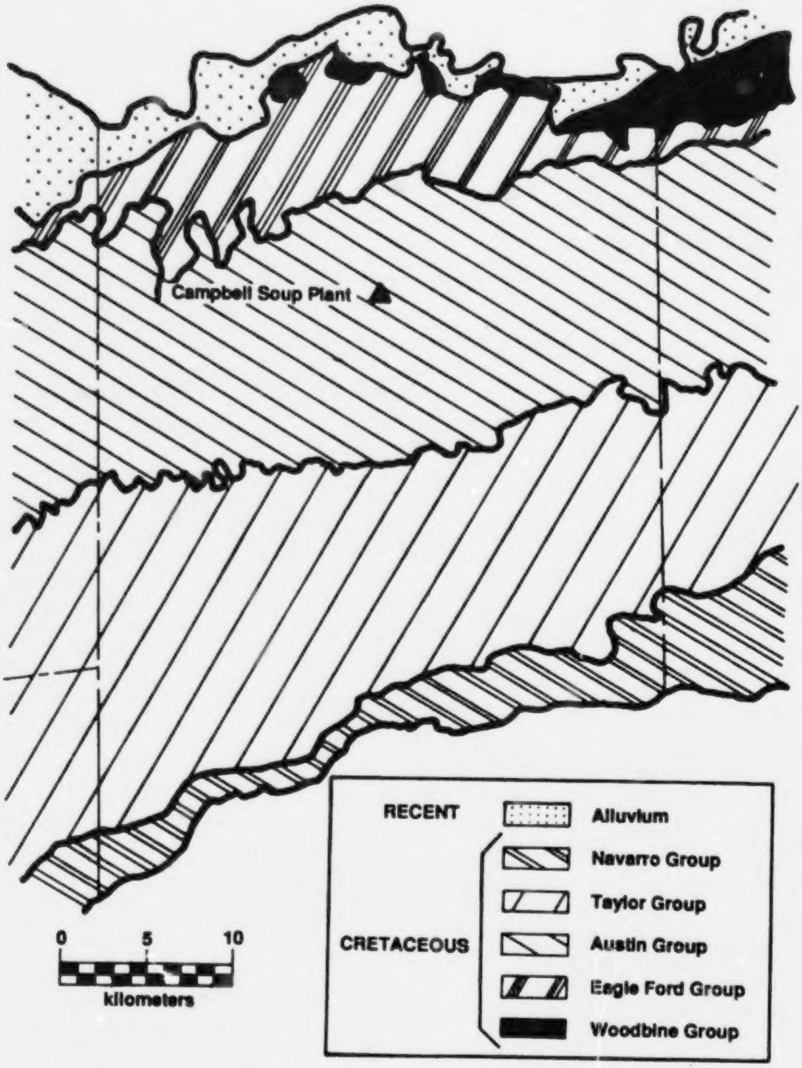

Campbell Soup Plant
RECENT
Alluvium
Navarro Group
Taylor Group
CRETACEOUS
Austin Group
Eagle Ford Group
Woodbine Group
0
5
10
kilometers

Figure 8-2 Geologic Outcrop Map: North-Central Texas

The principal source of recharge to the Blossom Sand is precipitation on the outcrop. The outcrop ranges in width from about 2.5-km near its western end in central Fannin County to about 10-km in central Red River County. Much of the outcrop in Red River County is covered by a mantle of thin high-level terrace deposits, which in places forms an excellent recharge facility for the Blossom Sand(22).

Water moves southward in the Blossom Sand from the recharge areas on the outcrop to points of discharge. The rate of movement is not known, but is expected to be very slow. Water is discharged from the Blossom by pumping from wells and by natural means, such as seepage into other formations in the subsurface, especially along the Luling-Mexia-Talco fault system (Figure 8-1).

The occurrence of useable quality water in the Blossom Sand is generally limited to the Red River basin and the northern part of the Sulfur River basin (Figure 8-3, Map of Aquifers in North-Central Texas). In general, groundwater from the Blossom Sand is fairly high in dissolved solids content (100 to 3,000 mg/L), high in sodium bicarbonate, and is soft(20).

Bonham Formation - Austin Group

The Bonham Formation of the Austin Group outcrops in central Fannin, Lamar and Red River Counties in north-central Texas. On the outcrop, the contact between the upper Eagle Ford and lower Austin (Bonham Formation) is usually considered unconformable. The nature of this unconformity in the subsurface, however, is not well known. The inferred hiatus (break or interruption in the continuity of a stratigraphic record) associated with the Eagle Ford-Austin unconformity appears to diminish towards the north-central portions of the basin(21).

The contact between the Bonham Formation and the Eagle Ford Group is an erosional surface with the upper surface of the Eagle Ford apparently eroded.

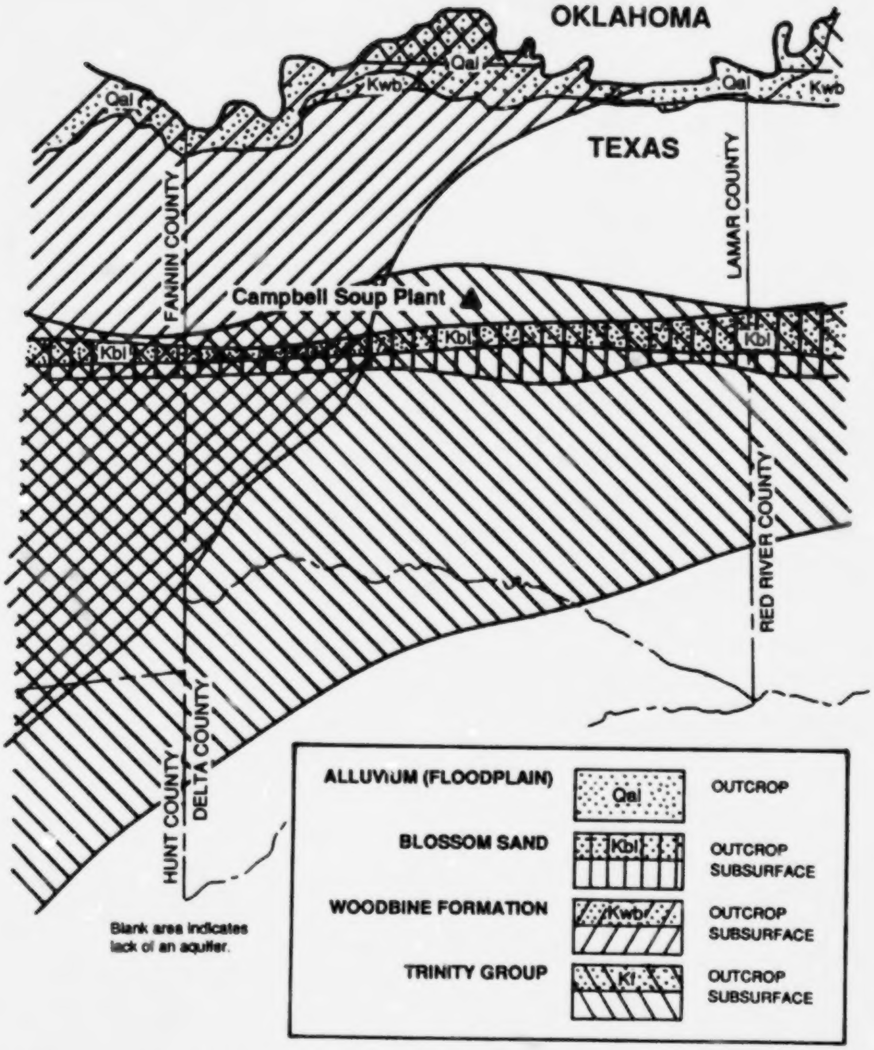

OKLAHOMA
TEXAS
Qal
Kwb
Qal
Qal
Kwb
FANNIN COUNTY
LAMAR COUNTY
Campbell Soup Plant
Kbl
Kbl
Kbl
RED RIVER COUNTY
HUNT COUNTY
DELTA COUNTY
Blank area indicates lack of an aquifer.
ALLUVIUM (FLOODPLAIN)
Qal
OUTCROP
BLOSSOM SAND
Kbl
OUTCROP
SUBSURFACE
WOODBINE FORMATION
Kwb
OUTCROP
SUBSURFACE
TRINITY GROUP
Kt
OUTCROP
SUBSURFACE

Figure 8-3. Map of Aquifers in North-Central Texas

The Bonham Formation consists of greenish gray to yellowish gray clay and marl, which becomes progressively more sandy to the east of Lamar County. Marine megafossils are common in the Bonham. A glauconite clay bed is found near the middle of the formation.

As with the Blossom sand, the Bonham formation strikes east-west and dips south to the Gulf. Thickness varies from 12 to 152-m thinning westward. The Bonham Formation is not known to yield useable water to wells in the north-central area.

Eagle Ford Group

The Eagle Ford Group of east Texas is a shale-dominated sequence containing localized deltaic (characterized by a delta) marine sands and consists predominantly of dark, bluish-gray, bituminous, laminated clays. The Eagle Ford varies in thickness from 60 to 275-m. It is thickest near the center of the basin and thins towards the southern and eastern margins of the basin. The Eagle Ford also thins eastward along the northern outcrop. While the dominant lithology is shale, the section contains interbeds of sandstone, limestone, and bentonite. Marine megafossils are common(21).

The Eagle Ford Group of the East Texas basin can be subdivided into four formations: the Tarrant, the Britton, the Arcadia Park, and the Sub-Clarksville Sand (Figure 8-4, Formations within the Eagle Ford Group)(21). These formations are mud-dominated with an increasing sand content upward in the section.

The Tarrant consists of interbedded sandstone and shale, which records the initial transgression of the Eaglefordian seas. The Britton consists of finely laminated highly organic clays which characterize Eagle Ford rocks of east Texas.

The Arcadia Park sediments are more clastic (shale) dominated and consist of 30 to 60-m of gray to dark gray, fissile (easily split), calcareous, mudstone with thin laminae of siltstone, sandstone, and fragmental limestone. In the northern outcrop region of the Eagle Ford Group, the Arcadia Park Formation is overlain by the Sub-Clarksville Formation of the Upper Eagle Ford strata (Figure 8-4). While the contact between the two is lithologically abrupt, it is conformable in nature(23).

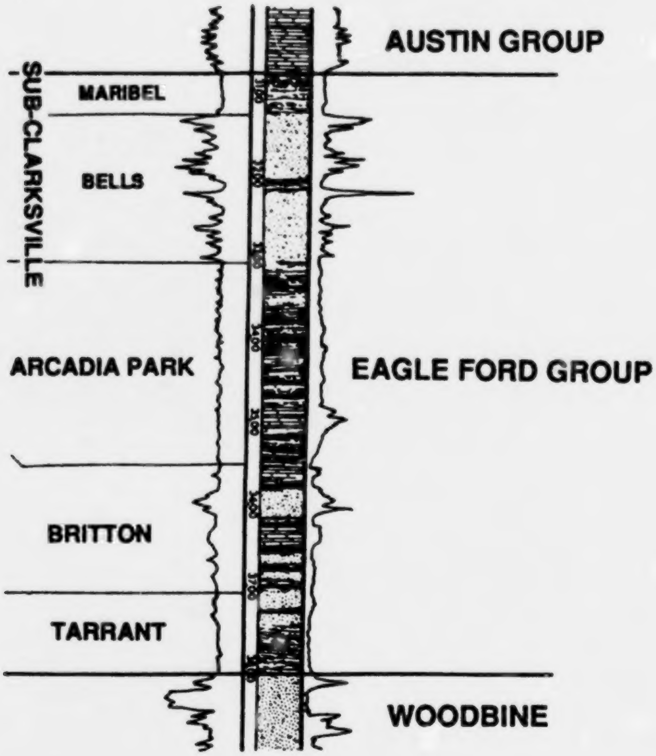

AUSTIN GROUP
SUB-CLARKSVILLE
MARIBEL
BELLS
ARCADIA PARK
EAGLE FORD GROUP
BRITTON
TARRANT
WOODBINE

Figure 8-4. Formations within the Eagle Ford Group

The Sub-Clarksville is the sand-dominated, upper-most unit of the Eagle Ford Group in the East Texas basin. Along the northern outcrop belt of the Eagle Ford Group the Sub-Clarksville is sub-divided into two members, a lower Bells Sandstone Member, and an upper Maribel Shale Member(21). The Bells Sandstone consists of gray to brown weathering quartz sandstone. The Maribel Shale consists of medium to dark gray laminated shale with silty partings, and thins eastward along the northern outcrop as the Bells thickens. The Maribel grades upward into a 1.5-m limestone bed that marks the top of the Eagle Ford Group.

The Eagle Ford Group is not considered a significant source of groundwater; however, shallow wells completed within the Eagle Ford in Lamar County yield small quantities of useable quality water(20).

MATERIALS AND METHODS

Monitoring Well Installation

Three monitoring wells were installed at at the Paris, Texas OLF site for collection of groundwater samples, water level measurement data, and hydraulic conductivity testing (slug tests).

The location of each well is indicated in Figure 8-5. A total of 2 open boreholes and 3 stainless steel cased wells were installed between 1987 and 1989. Installation and construction details were identical for all 3 wells and individual well logs and construction details are provided in Appendix A.

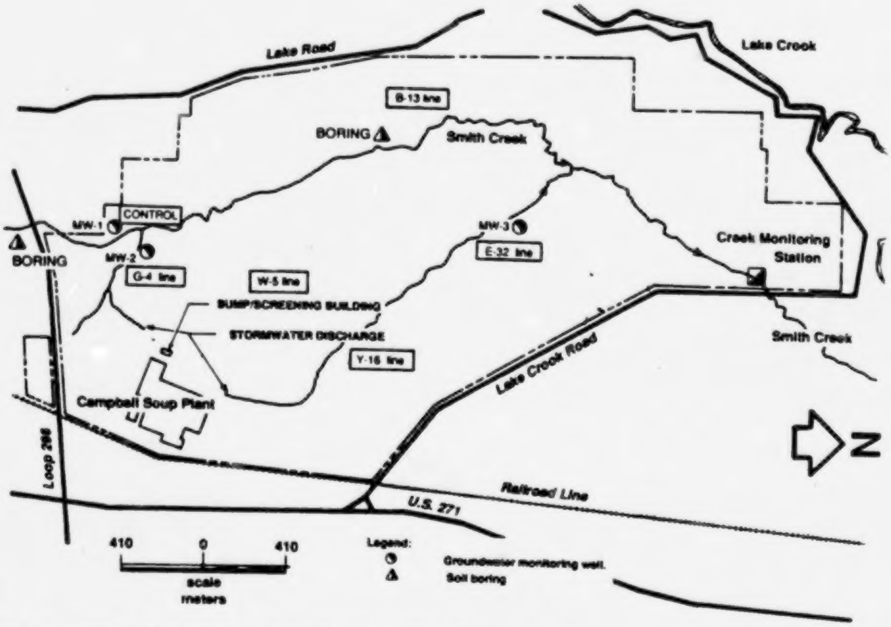

Lake Road
Lake Creek
B-13 line
BORING
Smith Creek
MW-1
CONTROL
BORING
MW-2
G-4 line
W-5 line
MW-3
E-32 line
SUMP/SCREENING BUILDING
STORMWATER DISCHARGE
Y-16 line
Creek Monitoring Station
Smith Creek
Lake Crook Road
Campbell Soup Plant
Loop 286
Railroad Line
U.S. 271
N
410
0
410
scale
meters
Legend:
Groundwater monitoring well.
Soil boring

Figure 8-5. Monitoring Well and Boring Location Map

The wells were drilled with a Central Mining Equipment hollow stem rotary auger. Generally, a solid core barrel was used during the drilling of MW-3. Core barrel samples, however, were collected from selected depths to monitor strata changes. A core barrel sampler was used exclusively during the installation of MW-1 and MW-2 in order to recover the full length of the bored soil (see Appendix A for details).

The wells were drilled to the depth at which the Austin-Eagle Ford contact was reached, between 5 and 8-m from the ground surface,. Although three wells were installed, five deep borings were actually completed. Two borings were made to depths in excess of 10-m (several meters deeper than the geologic contact). No evidence of groundwater was found at these two locations, and the boreholes were backfilled with cuttings and then abandoned (Figure 8-5). At each successfull borehole (MW-1 through MW-3), groundwater was found near or at the erosional contact between the Maribel Shale of the Eagle Ford Group and the Bonham Formation of the overlying Austin Group. The wells were screened within this zone.

The wells were cased with 5-cm diameter stainless steel (10-cm borehole diameter) and each was screened with a 1.5-m length (5-cm diameter) section of 0.25-mm slotted Johnson stainless steel screen. Washed sand was used for the filter pack and the sand filter extended between 0.3 and 0.6-m above and below the screen. The screen and filter pack were sealed off from the top of the casing with approximately 0.3-m of bentonite pellets. The annular space above the bentonite seal was packed with cuttings and the ground surface was sealed to a depth of approximately 1-m with Portland cement. Stainless steel locking caps were welded onto the top section of each monitoring well prior to installation and sampling.

Sampling of Monitoring Wells

Monitoring wells MW-1, MW-2 were sampled approximately twice yearly between 1987 and 1989 to identify the quality of groundwater at the Campbell Soup site. A total of 22 samples were collected on six dates at MW-1 and MW-2, i.e., 11 samples per well. A total of four samples were collected at MW-3 during 1988 and 1989.

Prior to sampling, each well was pumped until either three casing volumes had been removed or until each had been pumped dry several times. The wells were purged through Teflon tubing with a vacuum lift pump. Samples were collected by lowering a 3.8-cm diameter Teflon bailer into each well. The bailer was rinsed with distilled water between collections at different wells.

Each groundwater sample was analyzed for 20 purgeable organic compounds (volatile organics), 85 base/neutral extractables, 24 metals, TOC, ammonia, nitrate-nitrite, chlorides, sulfates, dissolved solids, alkalinity and pH. The latter two were determined in the field. A complete listing of the compounds and ions which were examined is provided in Appendix B.

Samples collected for volatile organic analyses were placed in borosilicate glass serum bottles, sealed, inverted and stored on ice in coolers. Samples for metals analyses were placed in 3.8-L brown, glass jugs and acidified to a pH of 2 with concentrated nitric acid. Samples scheduled for TOC, ammonia, and nitrate-nitrite analyses were placed in 1-L plastic cubitaners and acidified with 2-mL of concentrated sulfuric acid. All samples were collected in duplicate and placed on ice in coolers during transfer to RSKERL and UT-Austin laboratories, where the groundwater analyses were performed.

Slug Tests on Monitoring Wells

Slug tests were performed on two of the monitoring wells (MW-1 and MW-3) to measure the hydraulic conductivity of the water bearing zone around the well screens at the Austin-Eagle Ford contact. MW-2 was not tested because damage to the casing prevented the insertion of the slug testing equipment into the well. The slug test results were used to estimate the spatial variation of hydraulic conductivity within the Eagle Ford Formation-Austin Chalk contact at the Campbell Soup site. Combining the resulting hydraulic conductivity values with the porosity of the aquifer and slopes of the piezometric surface permited the prediction of pore water velocities and, hence, the rate of movement of groundwater contaminants.

The slug test involved "instantaneously" lowering (or raising) the water level in a well and measuring the water-level recovery through time. This technique was originally developed by Hvorslev(24) and has since been expanded to include a variety of field situations(25)(26)(27)(28). Although the results of this method are not precise(29) it is generally considered appropriate as a means of estimating the order of magnitude of field hydraulic conductivity(30)(28).

Several assumptions must be met for the method to be valid: (a) the change in water level in the well must be "instantaneous", so that no cone of depression is created around the well; this is necessary because the mathematical development of the slug test method(24). treats the head in the formation as a constant, and (b) flow to the well must be according to Darcy's Law. The well must be constructed so that neither the screen nor the filter pack inhibit or significantly enhance groundwater movement. Thus, a clogged screen or filter pack would render the well inappropriate for slug testing. In addition, the well cannot be developed to such an extent that a zone of increased permeability is developed around the well screen.

In performance of slug tests, the water level in the well is rapidly changed by removing water with a pump, bailer, or solid weight. The water level is then repeatedly measured at recorded times as recovery begins. Drawdowns are then normalized by dividing by the initial (maximum) drawdown. The normalized drawdowns are plotted versus time on a semilogarithmic scale; drawdowns are plotted on the logarithmic scale. Two values of drawdown on the linear portion of the curve and their corresponding time values are read from the graph and then substituted into the appropriate solution (based on well configuration) for the calculation of the hydraulic conductivity(26).

Slug tests were performed on MW-1 and MW-3 on July 18, 1989. Standing water was rapidly removed from the wells with a 3.8-cm diameter 1-L capacity PVC bailer. At the time when inflow equalled outflow (due to bailing) or the well was bailed dry, the slug test was begun. A stopwatch and a vinyl survey tape was used to measure the time dependent rate of water level rise. Each well was tested at least three times. The slug test results are listed in Appendix C.

Well Casing and Water Level Elevation Survey

A topographic survey was made at the site to identify the elevation and location of all monitoring wells and borings made at the Campbell Soup site. The data were used to determine groundwater flow directions and construct geologic cross sections of the site. The elevation survey was conducted by EPA personnel using standard survey practices.

RESULTS

Groundwater Hydrology

Water Level Analysis

Water level elevations were measured in the 3 monitoring wells on two dates in 1989. The water level measurements and ground surface elevations from July 18, 1989 and August 24, 1989 presented in Table 8-2.

TABLE 8-2. GROUND SURFACE AND WATER LEVEL ELEVATIONS*

Location	Ground Surface Elevation	Water Level, Potentiometric Surface**	Austin - Eagle Ford Contact
MW-1	152.26	151.73 / 151.73	148.01
MW-2	152.65	151.06 / 152.06	147.90
MW-3	142.51	141.75 / 142.68	135.37

* All elevations in units of meters above mean sea level.
** July 18,1989 / August 24,1989 measurements.

The flow direction of the groundwater below the Cambell Soup site was determined using these water level elevations. The general direction of flow in the aquifer was from the south to the north-west. The linear distance between MW-1 and MW-3 was 2,330-m; 200-m between MW-1 and MW-2; and 2,170-m between MW-2 and MW-3. Based on these data the hydralic gradient between MW-1 and MW-3 was calculated as 0.0039 m/m, and 0.0048 between MW-2 and MW-3.

A geologic cross section (Figure 8-6) of the subsurface between MW-1 and MW-3 was constructed using the data presented in Table 8-2 and the well logs (Appendix A). Because the elevation data represent only three locations over a linear surface distance of over

2,000-m, it was necessary to extrapolate a large part of the surface and subsurface contours between MW-2 and MW-3. The extrapolation was based on observations noted during the field investigations. The survey results indicate that although the regional dip of the Austin-Eagle Ford contact is to the south-east, the local dip is to the north-west. The difference in dip may be due to localized faulting or folding of the strata.

A rigorous analysis (including pumping tests and water level measurements at clustered piezometers) to determine the type of aquifer (i.e., confined, semi-confined, or unconfined aquifer) at the site was not conducted. However, based upon the available data, it was concluded that the aquifer could best be described as a either a semi-confined aquifer or a water table aquifer. Because the limited aquifer data exhibited characteristics of both of these systems it was not possible to definitively identify the aquifer type.

Drilling logs indicated that the overlying clay of the Bonham Formation was heavily fractured along the horizontal plane, thus it is possible that water may be transmitted relatively rapidly through these fractures in the less permeable clay and marl of the Bonham Formation to the highly permeable weathered contact zone between the Austin and Eagle Ford Groups. Therefore, although the entire soil system may be near or at saturation (i.e., a water table aquifer would exist) a well screened within the Bonham clay may not produce appreciable quantities of water unless the screen were to penetrate fractures which transmit significant quantities of groundwater. During well drilling no water was observed in the boreholes of MW-1, MW-2 and MW-3 until the contact zone was reached, at which point water rushed into the borehole and rose above or to within approximately a meter of the ground surface. Thus, upon initial inspection of field data , it may appear that the aquifer is confined. However, because of the high storativity and low transmissivity of the Bonham clays it is possible that a semi-confined aquifer exists at the site. These results were further supported by tensiomenter and neutron probe soil moisture measurements made at the site in 1968(8) which indicated that soil below the site was continuously near saturation.

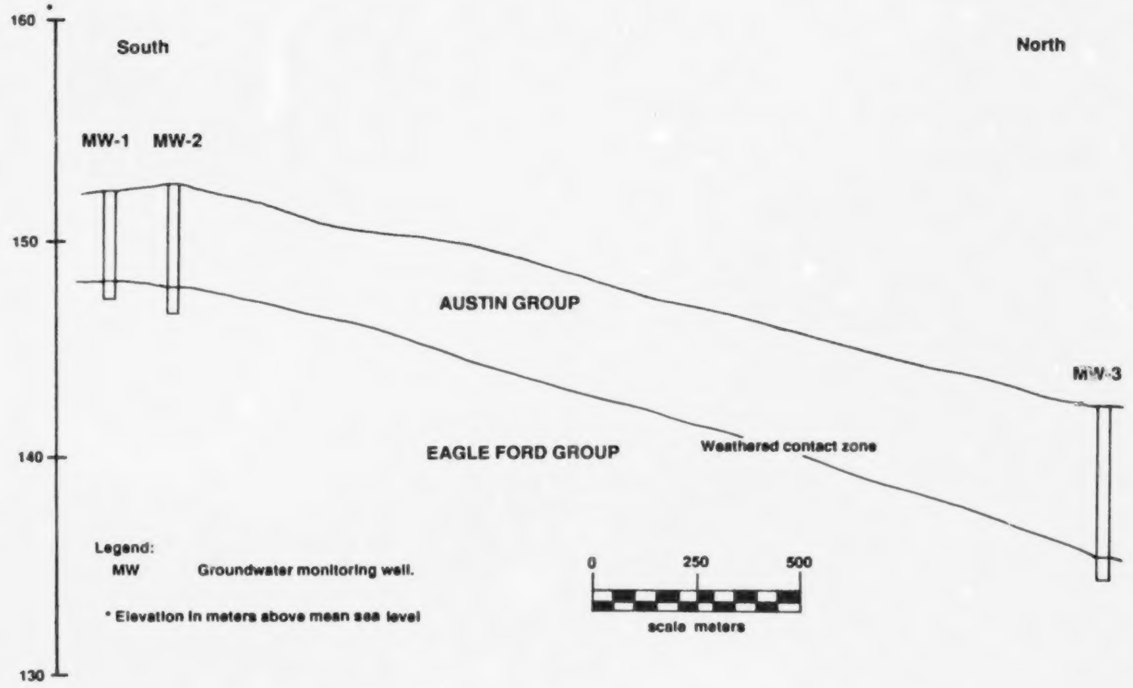

160
150
140
130
*
South
North
MW-1
MW-2
MW-3
AUSTIN GROUP
EAGLE FORD GROUP
Weathered contact zone
Legend:
MW Groundwater monitoring well.
* Elevation in meters above mean sea level
0
250
500
scale meters

Figure 8-6. Geologic Cross Section Between MW-1 and MW-3

Slug Tests

For the slug test analyses, the applicable well configuration equations(25) were taken for three possible cases. A cased well in a confined aquifer with (a) a partially penetrating well screen and, (b) a fully penetrating screen and, (c) a cased well with a partially penetrating screen in an unconfined aquifer. The appropriate equations for the partially penetrating case are:

Partially penetrating well screen: confined aquifer

$$k = \frac{\pi r^2}{C_s R (t_2 - t_1)} \ln\left(\frac{S_1}{S_2}\right) \qquad (8-1)$$

where:

k = hydraulic conductivity, L/T
π = pi, dimensionless
r = well radius, L
R = screen radius, L
t_x = time, T
S_x = drawdown at t_x, L
C_s = shape factor, dimensionless

$$C_s = \frac{2\pi (L/R)}{\ln(L/R) + 1.36} \qquad (8-2)$$

with:

L = screen length, L.

A complete listing of slug test data is provided in Appendix C. The calculated hydraulic conductivity for MW-1 (k = 1.5 x 10^{-4} cm/sec) was an order of magnitude less than for MW-3 (k = 3.5 x 10^{-3} cm/sec).

Because the type of aquifer and the exact location of the water zone (with respect to the well screen) was not known, the slug test calculations were performed for the two alternate cases of (a) a fully penetrating screen in a confined aquifer and, (b) a water table aquifer. These additional calculations were necessary to test the sensitivity of the calculated hydraulic conductivity to well configuration and aquifer type. The appropriate equations and their application are described by Cedergren(26). The results indicated that the calculated hydraulic conductivity was within the same order of magnitude, regardless of whether the screen was fully or partially penetrating or in an unconfined or confined aquifer. The results of the calculations are summarized in Table 8-3.

TABLE 8-3. SLUG TEST RESULTS FOR CONFINED AND UNCONFINED AQUIFERS

Well No.	Assumed Screen Geometry	Hydraulic Conductivity (cm/s)
MW-1	Partially penetrating*	1.5×10^{-4}
MW-1	Fully penetrating**	2.3×10^{-4}
MW-3	Partially penetrating*	3.5×10^{-3}
MW-3	Fully penetrating**	5.5×10^{-3}

* Confined and unconfined aquifer conditions
** Confined aquifer conditions only

Calculated hydraulic conductivity values for the partially penetrating confined case and the unconfined case were identical. The calculated hydraulic conductivity values foe all cases were substantially greater than typical values for unweathered marine clay and shale (k between 10^{-11} and 10^{-8} cm/s). The calculated values, however, were well within the range of hydraulic conductivities reported for fractured igneous and metamorphic rocks as well as clean sand (k between 10^{-4} and 10^{-3} cm/s)(6).

These results indicate that fractures in the Mirabel Shale and sandy laminae at the weathered contact zone between the Eagle Ford Group and the Austin Chalk represent a path for the rapid movement of groundwater at the Paris, Texas site.

CONCLUSIONS

The Paris, Texas area is underlain by hundreds of meters of sediments laid down from the Paleozoic through the Quarternary eras. A continuing cycle of advance and retreat of ancient seas resulted in periods of sediment accumulation alternating with periods of erosion. The Austin Chalk Group outcrops at the Campbell Soup (Texas) Inc. OLF site. The Bonham Formation, within the Austin Group, is a heavy clay with marl, about 6 to 15-m thick at the site. The Bonham Formation is underlain by the Eagle Ford Group. The Eagle Ford Group is composed primarily of dark, fissile shale. The contact between the Eagle Ford and Austin Groups is characterized by an erosional surface and is unconformable.

The results of the hydrogeologic investigation indicated that an aquifer exists below the OLF site. Data limitations prevented the definitive classification of the aquifer, i.e., either semi-confined, confined or unconfined. The available data, however, suggest that the aquifer is semi-confined and that the erosional contact between the Eagle Ford and the Austin Groups may serve as a significant transmission zone for groundwater within the relatively impermeable clays of the Bonham Formation.

Groundwater level data indicate that the general flow direction was to the north-west with a hydraulic gradient of approximately 0.004 m/m. Measured hydraulic conductivity values within this zone (k between 10^{-4} and 10^{-3} cm/s) were substantially greater than typical values for unweathered marine clay and shale (k between 10^{-11} and 10^{-8} cm/s). However, the calculated values were well within the range of hydraulic conductivities reported for fractured rocks as well as for clean sand (k between 10^{-4} and 10^{-3} cm/s)(6).

These results indicate that fractures in the Mirabel Shale and sandy laminae at the weathered contact zone between the Eagle Ford Group and the Austin Chalk represent a path for the relatively rapid movement of groundwater at the Paris, Texas site.

CHAPTER 9

GROUNDWATER: QUALITY, RECHARGE, AND GEOCHEMISTRY

INTRODUCTION

The assessment of the long-term effects of OLF treatment on groundwater quality required the collection of "background" and possibly "impacted" groundwater samples. This section presents groundwater quality data collected at the Paris, Texas OLF site between 1987 and 1989.

GROUNDWATER QUALITY DATA

Monitoring wells MW-1, MW-2 were sampled approximately twice yearly between 1987 and 1989 to identify the quality of groundwater located within the erosional contact of the Austin and Eagle Ford Groups at the Campbell Soup site. A description of the monitoring well installation is given in Chapter 8. A total of 22 samples were analyzed. The samples were collected on six dates at MW-1 and MW-2, i.e., 11 samples per well. A total of four samples from MW-3 were analyzed. The samples were analyzed for the constituents summarized in Table 3-1. A detailed list of the constituents is provided in Appendix B.

With the exception of the samples collected on 6/17/87, all samples were analyzed in duplicate; i.e., one sample was filtered through a 0.45 micron filter while the other was not filtered prior to analysis. The samples were filtered to remove clay particles which can become suspended in the groundwater during the process of sample collection. The inadvertent suspension of clay particles may result in the increase in the concentrations of clay constituents such as aluminum, iron, and magnesium and any chemicals sorbed to the clay, in the unfiltered, digested sample over the filtered, digested sample.

The groundwater monitoring data for constituents detected in the three wells are presented in Tables 9-1, 9-2, and 9-3. No extractable organic compounds and purgeable organic compounds (as listed in Appendix B) were detected in the groundwater from any of the wells at any time. Therefore, these constituents were not included in the Tables 9-1 through 9-3.

Based on the general classification of Winslow and Kister(31), the groundwater samples from all three wells were moderate to very saline, with TDS values between 7,000 and 13,000 mg/L. The salinity was primarily due to the presence of the following dissolved ions: chloride, sulfate, sodium, calcium, and magnesium.

TOC values were low (<10 mg/L) at all wells and this condition is typical for a groundwater which has not be subjected to contamination by organic material.

Water samples were collected at MW-1 and MW-2 on the following dates: June 11, 1987; June 17, 1987; September 3, 1987; June 1, 1988; November 1, 1988; and April 4, 1989. Groundwater data from MW-1 (over time) for the major cations present (calcium, magnesium and sodium) with standard deviations are presented in Figures 9-1 through 9-3. The figures illustrate the consistency of the groundwater quality at MW-1. Similar trends for these constituents were observed at MW-2. Means and standard deviation values for the major anions present (chloride and sulfate) could not be calculated because duplicate analyses for these constituents were not performed.

Two samples were collected at MW-3 on November 1, 1988 and April 5, 1989; consequently, insufficient data existed to analyze the data with respect to time for that well.

The data from MW-1 and MW-2 were tested by one-way analysis of variance (ANOVA) to determine if the means with respect to sampling date and location were representative of long-term and location water quality. Mean values were desired to provide a rapid means for comparison of water quality between locations.

TABLE 9-1. GROUNDWATER CONSTIUENTS DETECTED[a] AT MW-1: 1987 THROUGH 1989

Constituent	6/11/87*	6/11/87**	6/17/87*	9/3/87*	9/3/87**	6/1/88*	6/1/88**	11/1/88*	11/1/88**	4/5/89*	4/5/89**
Inorganic Cations											
Aluminum	24	0.4	8.2	0.4	<0.1	0.96	<0.1	6.44	<0.1	5.64	6.44
Barium	0.24	0.07	0.10	0.03	0.02	0.02	0.02	0.09	0.01	0.07	0.09
Boron	2.6	1.8	2.7	2.6	2.3	0.55	0.59	2.89	2.99	2.10	2.89
Calcium	400	450	400	530	510	485	499	464	468	491	464
Chromium	0.03	<0.02	<0.01	<0.01	<0.01	0.02	<0.01	0.11	<0.01	0.07	0.11
Cobalt	0.01	<0.02	0.01	0.01	0.01	<0.01	<0.01	0.01	0.01	0.01	0.01
Iron	20	0.4	6.6	2.5	1.9	3.22	1.88	10.0	1.90	7.13	0.49
Lithium	0.34	0.42	0.35	0.38	0.36	0.23	0.29	0.37	0.37	0.44	0.42
Magnesium	240	250	230	280	260	335	340	173	174	250	241
Manganese	0.60	0.52	0.58	0.72	0.68	0.42	0.42	0.52	0.53	0.39	0.38
Nickel	0.01	<0.02	0.01	<0.01	<0.01	0.05	0.05	0.03	0.02	0.04	0.04
Potassium	8.7	9.7	7.6	6.7	6.4	5.3	5.52	10.05	9.95	8.58	8.00
Sodium	2,600	2,700	2,900	3,300	3,000	3,510	3,690	2,420	2,240	2,490	2,550
Strontium	9.8	12	9.7	11	11	11.6	11.7	9.16	9.11	10.4	10.1
Thallium	0.02	<0.05	0.03	<0.02	<0.02	<0.02	<0.02	0.06	0.07	0.02	0.04
Titanium	0.6	<0.2	0.2	<0.1	<0.1	<0.1	<0.1	0.29	0.11	0.29	<0.1
Zinc	0.03	0.41	<0.01	<0.01	<0.01	0.01	<0.01	0.01	0.03	0.01	<0.01
Inorganic Anions											
Chloride	2,419	NA	2,200	2,334	NA	2,200	NA	1,600	NA	NA	NA
Sulfate	5,025	NA	5,075	5,740	NA	6,360	NA	4,140	NA	NA	NA
Additional Analyses											
Alkalinity	600	NA	660	600	NA	640	NA	580	NA	NA	NA
Ammonia	1.0	NA	0.9	1.0	NA	0.7	NA	NA	NA	NA	NA
NO_3-NO_2-N	<0.05	NA	0.3	<0.05	NA	<0.05	NA	NA	NA	NA	NA
pH	NA	NA	NA	7.2	NA	6.89	NA	NA	NA	NA	NA
TOC	14	NA	<2	NA	NA	2.6	NA	NA	NA	NA	NA

[a] All values in units of mg/L except pH, and alkalinity in mg/L as $CaCO_3$.

* Sample was not filtered prior to analysis.

** Sample was filtered through 0.45 micron filter prior to analysis.

NA Not analyzed.

TABLE 9-2. GROUNDWATER CONSTITUENTS DETECTED[a] AT MW-2: 1987 THROUGH 1989

Constituent	6/11/87*	6/11/87**	6/17/87*	9/3/87*	9/3/87**	6/1/88*	6/1/88**	11/1/88*	11/1/88**	4/5/89*	4/5/89**
Inorganic Cations											
Aluminum	3.3	<0.1	2.0	0.2	<0.1	0.63	<0.1	1.44	<0.1	4.68	<0.1
Barium	0.06	0.05	0.04	0.03	0.03	0.04	0.04	0.02	0.01	0.04	0.01
Boron	1.0	1.0	1.1	1.0	1.3	0.58	0.57	1.22	1.32	0.92	0.89
Calcium	520	520	520	600	570	516	521	503	515	504	499
Chromium	<0.01	<0.01	<0.01	<0.01	<0.01	0.03	<0.01	<0.01	<0.01	0.02	<0.01
Cobalt	<0.01	0.01	0.01	0.01	0.02	0.01	0.01	0.01	0.01	0.01	0.01
Iron	2.5	<0.1	1.7	0.3	0.2	1.21	0.27	2.53	0.36	4.41	0.15
Lithium	0.70	0.69	0.63	0.80	0.53	0.87	0.77	0.58	0.58	0.85	0.66
Magnesium	110	110	110	130	120	126	127	110	111	125	113
Manganese	1.6	1.6	1.7	1.9	2.2	1.91	1.93	1.75	0.36	1.72	1.66
Nickel	0.01	0.01	0.01	0.03	0.02	0.02	0.02	0.03	0.02	0.04	0.03
Potassium	8.6	8.2	7.7	8.3	7.4	7.16	8.06	10.9	11.2	11.2	10.12
Sodium	1,700	1,700	1,700	2,100	2,000	2,110	1,990	1,750	1,950	1,740	1,790
Strontium	11	11	11	12	11	12.9	12.6	11.4	11.4	12.1	10.9
Thallium	0.03	<0.02	0.03	<0.02	<0.02	<0.02	<0.02	0.06	0.09	0.02	0.06
Titanium	0.1	<0.1	0.1	<0.1	<0.1	<0.1	<0.1	0.20	0.12	0.26	<0.1
Zinc	<0.01	0.21	<0.01	<0.01	<0.01	0.03	0.02	0.01	0.01	0.01	0.01
Inorganic Anions											
Chloride	1,629	NA	1,479	1,523	NA	1,500	NA	2,450	NA	NA	NA
Sulfate	3,450	NA	3,425	3,510	NA	3,500	NA	3,320	NA	NA	NA
Additional Analyses											
Alkalinity	NA	NA	NA	NA	NA	NA	NA	650	NA	NA	NA
Ammonia	NA	NA	0.5	1.0	NA	0.4	NA	1.40	NA	NA	NA
NO_3-NO_2-N	NA	NA	0.3	<0.05	NA	<0.05	NA	<0.05	NA	NA	NA
pH	NA	NA	6.9	7.02	NA	6.82	NA	6.97	NA	NA	NA
TOC	NA	NA	<2.0	<2.0	NA	5.0	NA	4.7	NA	NA	NA

[a] All values in units of mg/L except pH, and alkalinity in mg/L as $CaCO_3$.

* Sample was not filtered prior to analysis.

** Sample was filtered through 0.45 micron filter prior to analysis.

NA Not analyzed.

TABLE 9-3. GROUNDWATER CONSTITUENTS DETECTED[#] AT MW-3: 1988 THROUGH 1989

Constituent	11/1/88[+]	11/1/88[++]	4/5/89[+]	4/5/89[++]
Inorganic Cations				
Aluminum	10.2	9.51	12.3	11.5
Barium	0.03	0.02	0.01	<0.01
Boron	1.49	1.53	1.68	<0.03
Calcium	491	505	494	512
Chromium	<0.01	<0.01	<0.01	<0.01
Cobalt	0.23	0.23	0.24	0.25
Iron	4.51	4.12	4.32	4.32
Lithium	0.77	0.77	0.95	0.07
Magnesium	140	141	152	153
Manganese	4.00	4.11	4.28	4.40
Nickel	0.35	0.35	0.35	0.35
Potassium	15.2	15.0	16.0	15.2
Sodium	1,240	1,420	1,280	1,350
Strontium	8.31	8.49	9.14	<0.01
Thallium	0.06	0.11	0.03	0.07
Titanium	0.13	0.12	<0.1	<0.1
Zinc	1.92	1.51	1.47	1.54
Inorganic Anions				
Chloride	1,200	NA	NA	NA
Sulfate	3,100	NA	NA	NA
Additional Analyses				
Alkalinity	34	NA	NA	NA
Ammonia	0.40	NA	NA	NA
NO_3-NO_2-N	<0.05	NA	NA	NA
pH	4.9	NA	5.4	NA
TOC	6.7	NA	NA	NA

[#] All values in units of mg/L except pH, and alkalinity as mg/l $CaCO_3$.

[+] Sample was not filtered prior to analysis.

[++] Sample was filtered through 0.45 micron filter prior to analysis.

NA Not analyzed.

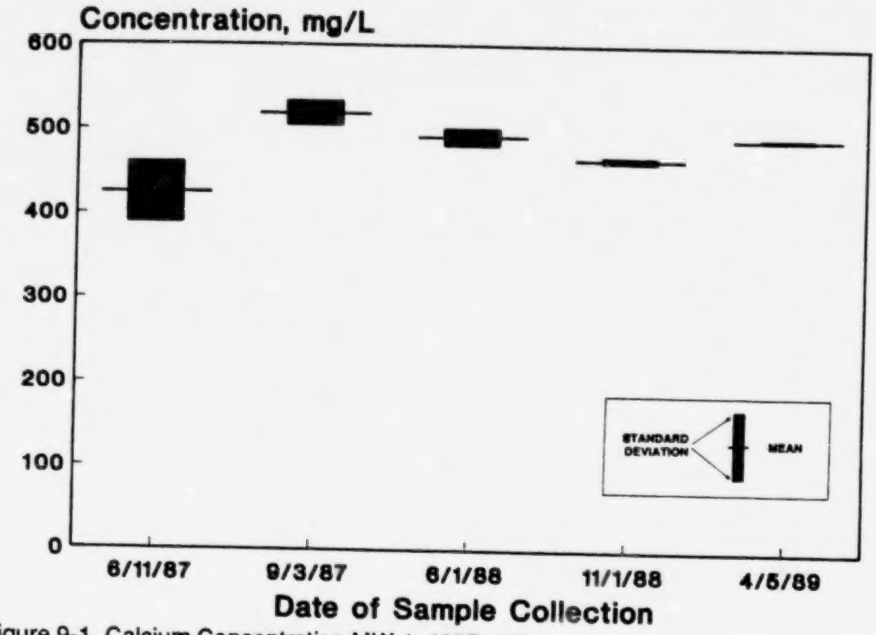


Figure 9-1. Calcium Concentration MW-1: 1987-1989

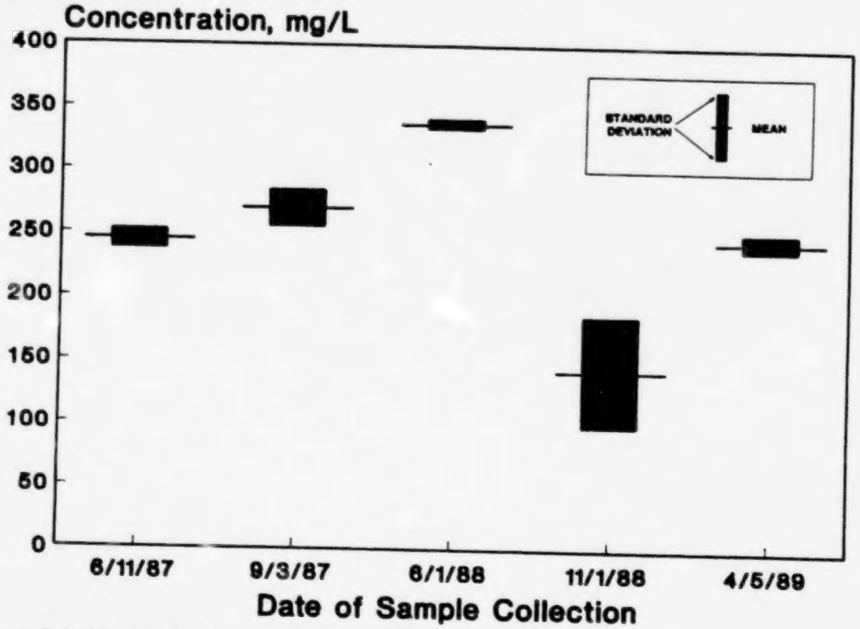


Figure 9-2. Magnesium Concentration MW-1: 1987-1989

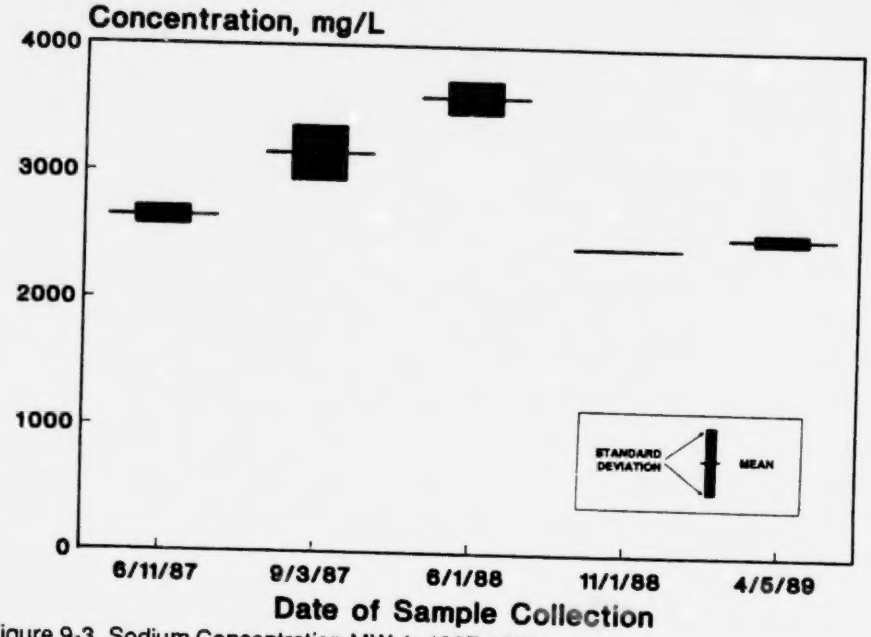


Figure 9-3. Sodium Concentration MW-1: 1987-1989

The ANOVA tests were applied to potassium, calcium, magnesium, manganese, sodium, chloride, and sulfate groundwater concentration data. These ions were selected for ANOVA analyses because: (a) they represented the major cations and anions present in groundwater as well as soil, (b) soil data indicated that leaching of several of these ions may have occurred, and (c) duplicate samples of these ions, with the exception of chloride and sulfate analyses, were available from 5 of the 6 sampling events.

Analysis of the soil data indicated that nickel and zinc had accumulated in the wastewater appliction areas. Therefore, the groundwater data was examined to determine if there was any effect on water quality due to the presence of these two metals in the surface soil at the site. Nickel and zinc concentrations in all samples collected at MW-1 and MW-2 were near or below instrument detection limits (0.01 mg/L). However, significantly greater concentrations of zinc (mean = 1.61 mg/L) and nickel (mean = 0.35 mg/L) were detected in all

4 samples from MW-3. The higher zinc and nickel concentrations at MW-3 may have been due to increased solublization of these elements in the groundwater because of the low pH (4.9) in the vicinity of the well.

Because of error inherent in sample measurement when the concentration is near the instrument detection limit, the nickel and zinc data were excluded from the ANOVA test. The results of the ANOVA testing ($p \le 0.05$) on the calcium, magnesium, manganese, potassium, sodium, chloride, and sulfate data collected at MW-1 and MW-2 are given in Table 9-4. Data from MW-3 were not included due to the limited size of the data set available for that well.

TABLE 9-4. GROUNDWATER MONITORING WELL DATA (MW-1 & MW-2) ANOVA^{+} TEST RESULTS

Constituent	Well Location	Collection Date
Calcium	${}^{1}_{18}F = 9.75$	not significant
Magnesium	${}^{1}_{18}F = 35.8$	not significant
Manganese	${}^{1}_{18}F = 50.8$	not significant
Potassium	not significant	${}^{4}_{15}F = 10$
Sodium	${}^{1}_{18}F = 38.3$	not significant
Chloride	not significant	not significant
Sulfate	${}^{1}_{6}F = 15.22$	not significant

$^{+}$ Critical F values: ${}^{4}_{15}F = 3.06$; ${}^{1}_{18}F = 4.41$; ${}^{1}_{15}F = 4.54$; ${}^{1}_{6}F = 5.99$

The ANOVA testing indicated that, with respect to location, statistically significant differences between the means were present. However, for all metals and anions tested by ANOVA, with exception of potassium, there was no statistical difference in groundwater quality over the time period of study (June 1987 through April 1989). Thus, mean concentration values at each location (with respect to time) of groundwater quality indicies were suitable for use in comparisons and summarizations.

Because insufficient data were available to test for equality of the mean concentration values measured at MW-3 it was assumed that water quality indicators at MW-3 followed the same trends (with respect to location and sampling period) as at MW-1 and MW-2.

Therefore, mean values were calculated for the data obtained at MW-3. Likewise, mean values were calculated for nickel and zinc (for MW-1, MW-2, and MW-3) although the data were deemed inappropriate for ANOVA testing.

The mean values were adequate for simple, rapid comparisons of water quality between the wells. A summary of the groundwater monitoring data (mean values) for all three wells is given in Table 9-5.

TABLE 9-5. GROUNDWATER QUALITY[*] AT CAMPBELL SOUP SITE: MEAN[#] VALUES AND STANDARD DEVIATION

Constituent	MW-1	MW-2	MW-3
Calcium	**472** ± 42	**525** ± 32	**496** ± 6.5
Magnesium	**247** ± 65	**129** ± 38	**169** ± 49
Manganese	**0.52** ± 0.11	**1.79** ± 0.18	**4.20** ± 0.18
Nickel	**0.03** ± 0.02	**0.03** ± 0.02	**0.35** ± 0.0
Potassium	**7.9** ± 1.7	**8.8** ± 1.55	**13.5** ± 3.7
Sodium	**2,870** ± 452	**2,026** ± 415	**1,623** ± 623
Zinc	**0.02** ± 0.01	**0.02** ± 0.01	**1.61** ± 0.21
Chloride	**2,151** ± 322	**1,856** ± 440	1,200 [#]
Sulfate	**5,268** ± 834	**3,441** ± 76	3,100 [#]
Alkalinity as $CaCO_3$	**610** ± 42	**655** ± 7	**27** ± 10
TOC	**<10**	**<10**	NA
pH[**]	6.9 - 7.2	6.8 - 7.0	4.9 - 5.4

[*] All units in mg/L, except pH.

[#] Mean values in boldface, standard deviation in standard type.

[##] Single analysis performed.

[**] Range provided for pH.

DATA ANALYSIS AND DISCUSSION

Background Water Quality

Water quality data obtained from wells near the Campbell Soup plant and screened within the same aquifer and geologic formation can be used as a measure of "background" groundwater quality. However, inspection of well records for Lamar County(20) indicated that there were no wells within the immediate vicinity of the Paris, Texas area (as well as the entire County) which were screened within the Austin Chalk-Eagle Ford Group contact. The absence of wells at the contact may be attributed to the low probability of obtaining useable quantities of water from the contact zone (Chapter 8).

The presence of springs within the Bonham clay near the site and the quality of the spring waters can provide an indication of water quality which may be expected from wells screened within the Bonham Formation. However, field reconnaissance conducted as part of this study and an examination of records of springs in Lamar County and adjacent Counties(32) indicated that no springs discharged from the Bonham clays at or near the site. Most springs in Lamar County discharged from the Blossom sand and the Woodbine Formations. The spring waters were classified as sodium bicarbonate type, usually fresh, soft to moderately hard, and of neutral pH.(32) These findings indicated that spring water quality in the area could not be used as an indicator of background groundwater quality at the OLF site because the springs do not develop within nor do they discharge from the Bonham Formation.

Research conducted at the Paris, Texas OLF site in 1968 included an examination of soil water quality at several wastewater application locations.(10) The ground water samples were obtained from ceramic, porous cup lysimeters set at approximately 1-m below the ground surface. Lysimeters were located in the G-section of the system along lines which, at the time of sample collection (April 1968), had received wastewater for 2-years (G-11) and 4-years (G-4). The lysimeter data are presented in Table 9-6.

Because MW-2 was also located along the G-4 wastewater spray line, the lysimeter data from 1968 were used in direct comparisons of groundwater quality after 2, 4, and 24-years of wastewater application. The comparison is presented in the following subsection.

TABLE 9-6. LYSIMETER DATA*: 1-METER BELOW GROUND SURFACE, APRIL 1968

Constituent	G-4 line**	G-11 line***
Calcium	88	24
Magnesium	20	2
Potassium	1.4	0.6
Sodium	257	30
Chloride	190	69
Sulfate	570	25
Alkalinity as $CaCO_3$	60	30
pH*	6.7	6.0

* All units in mg/L, except pH.

** 4-years of wastewater application at collection.

*** 2-years of wastewater application at collection.

Groundwater Seepage Quantities

Darcy's law can be used to estimate seepage (aquifer recharge) through saturated and unsaturated soils if the hydraulic gradient, hydraulic conductivity, and wetted area are known or can be estimated with reasonable accuracy. Darcy's law is commonly stated as:

$$q = kiA \qquad (9-1)$$

with q equal to the quantity of seepage, k the hydraulic conductivity, i the hydraulic gradient, dh/dl (change in hydraulic head/change in length) and A, the cross sectional area.

The groundwater recharge process at the Paris, Texas OLF site can be conceptually described using Darcy's law as follows. Approximately 365 ha (3,650,000 m^2) were irrigated at the OLF site. Previous research at the site(10) indicated that the saturated surface and moist subsurface clay soils had a hydraulic conductivity of approximately 10^{-6} cm/sec. If it is also assumed that (a) the depth of penetration of saturation, l, just equals the hydraulic head, h, that is forcing water downward into the soil and (b) capillarity is neglected, then the hydraulic gradient, i, is equal to unity. Hence, for the assumed conditions and applying Darcy's law, the soil at the wastewater application areas can accept approximately 3,150 m^3/d, or 20% of the mean daily applied flow of 16,000 m^3/d. This calculated infiltration percentage agrees with the mean estimated infiltration percentage (20%) reported in earlier studies at the Paris, Texas site(10).

If it is assumed that wastewater application occurs 350 days per year and the infiltration rate remains constant, over 1,104,000 m^3/yr (292 million gallons) of water would theoretically seep through the clay soil at the site over a one-year period. Over the course of 24-years, this could result in the addition of over 26,500,000 m^3 (7 billion gallons) of water to the subsurface. In addition, the simple application of Darcy's law indicates that water would move vertically downward through the saturated clays at a rate of approximately 1-m per year.

Using a daily recharge rate of 3,150 m^3/d, a simple calculation was made to estimate the time in years required to saturate the clay underlying the OLF site, and thus, develop an aquifer where one did not exist previously. The calculations were made with the following assumptions: (a) annual recharge = 1,104,000 m^3/yr, (b) mean saturated thickness of 7-m, (c) wetted surface area of 365-ha and, (d) and an effective porosity of 0.35. The effective porosity is the fraction of the total volume of clay that consists of interconnected pores. The effective porosity corresponds to the volume of void space which must be filled with water to establish saturated conditions. The selection of 365-ha represented the most conservative scenario because in the 1960's and early 1970's less than half of the 365-ha were used for wastewater application, yet the volume of water applied and presumably subject to recharge was about the same (i.e., annual recharge = 1,104,000 m^3/yr). The mean saturated thickness of 7-m represented the case in which the semi-confined aquifer located at the Austin

Chalk-Eagle Ford contact (approximate aquifer thickness < 1-m) would be slowly recharged due to high porewater pressures in the overlying clay of the Bonham Formation, which would be near or at saturation.

Based on these assumptions, a semi-confined aquifer would be formed after 8 years of wastewater application. The aquifer would be semi-confined from above by clays near or at saturation with a uniform thickness of 7-m. The selection of a more conservative value of the effective porosity ($n_e = 0.1$), would result in the development of an aquifer after only 2 years of wastewater application. Although several simplifying assumptions were made to proceed with the calculations, the results indicated that the estimated recharge of treated wastewater to the subsurface would be more than sufficient to develop and maintain (within a relatively short period of time) saturated conditions in the upper confining unit and a semi-confined aquifer below the entire OLF site.

Hydrochemical Facies

Hydrochemical facies are distinct zones that have cation and anion concentrations describable within defined composition categories(6). The definition of a composition category is commonly based on the subdivisions of a trilinear diagram (Figure 9-4). The trilinear diagram represents a percentage plotting of cations and anion concentrations (meq/L) in separate triangles.

Hydrochemical facies are used to describe bodies of groundwater in an aquifer which differ in chemical composition(33). The facies are a function of the lithology, solution kinetics and flow patterns of the aquifer. Hydrochemical facies can provide a simple and rapid means of identifying different waters at one or many locations. Using this system (Figure 9-4), sulfate-chloride facies were dominant for groundwater collected at all three monitoring wells (MW-1, MW-2 and MW-3) and at the G-4 lysimeter (in 1968).

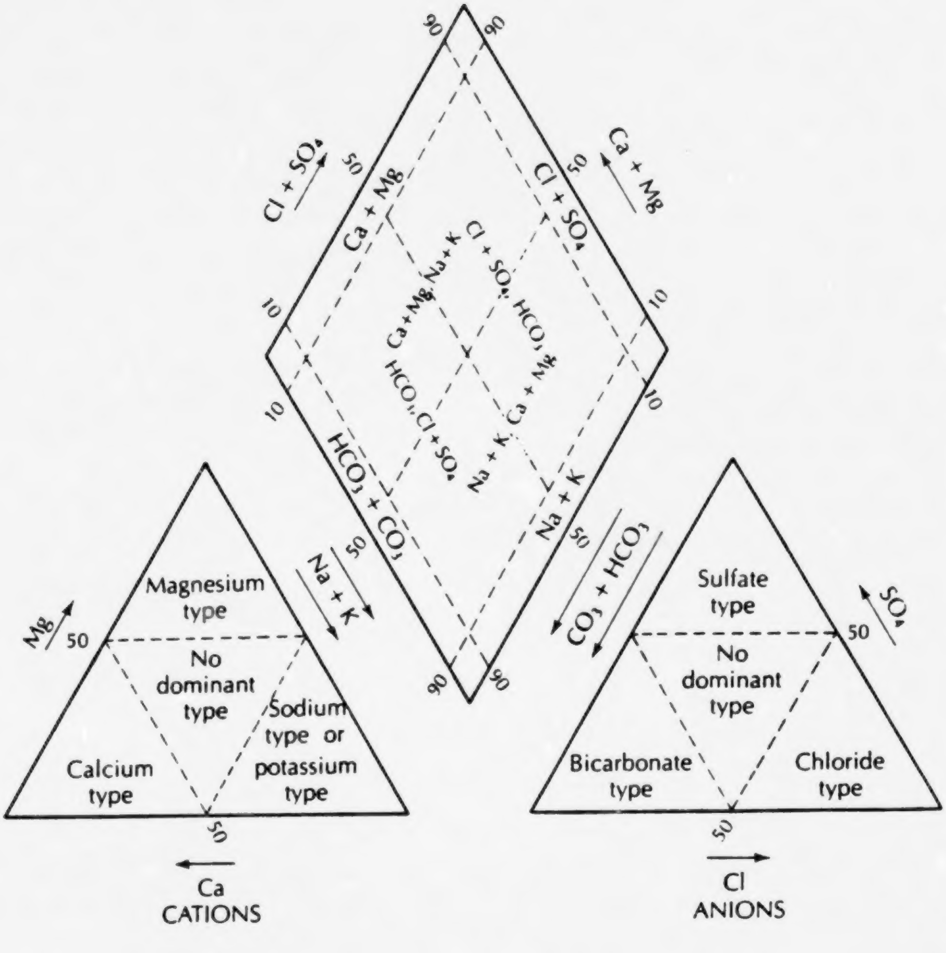

Cl + SO4
Ca + Mg
Ca + Mg
Cl + SO4
Na + K
Cl + SO4
HCO3
Ca+Mg
HCO3
Cl+SO4
Na + K
Ca + Mg
HCO3 + CO3
Na + K
90
90
50
50
10
10
10
10
50
50
90
90
Mg
50
Magnesium type
No dominant type
Sodium type or potassium type
Calcium type
Na + K
50
Ca
CATIONS
CO3 + HCO3
Sulfate type
No dominant type
Bicarbonate type
Chloride type
SO4
50
50
Cl
ANIONS

Figure 9-4. Trilinear Diagram Classification of Anion and Cation Facies

The use of hydrochemical facies provided further evidence of the similarity of water quality and geochemical evolution between the water samples collected at the three monitoring wells and the G-4 line lysimeter. The results suggest that groundwater characteristics below the site are the result of the dissolution of naturally present soluble minerals which contain sulfate and chloride as the predominant ions. These ions would be expected in the Bonham clays based upon the geologic history of the site (Chapter 7) and the results of the soil core sample analyses (Chapter 8).

Major-Ion Aqueous Geochemistry

Schoeller diagrams provide a convenient means for the visual inspection of chemical data from groundwater samples. The Schoeller semi-logarithmic plot permits the total concentration (milliequivalents/L) of major cations and anions of several water samples to be represented on a single graph in which major groupings or trends in the data can be discerned visually(6). The ionic composition lines for mean concentrations (meq/L) of calcium, magnesium, sodium and potassium, chloride, sulfate and bicarbonate in groundwater samples collected from all three wells (MW-1, MW-2, and MW-3) are plotted in Figure 9-5. Mean values were used because, as described above, there was no statistically significant difference in the mean concentration values of these major ions over time at each well.

Bicarbonate values were calculated using ionic concentration data for the major constituents with the relationship:

$$(Na^{+}) + (Mg^{2+}) + (Ca^{2+}) = (HCO_3^{-}) + (SO_4^{2-}) + (Cl^{-}) \qquad (9\text{-}2)$$

where ( ) represent units of milliequivalents per liter.

The similarity among the ionic composition lines on the Schoeller plot suggested a "match" between the water at the three wells. These results indicate that although groundwater quality varied among the three locations, the waters have apparently undergone similar patterns of geochemical evolution. Further, the results suggest a weak trend of greater ionic concentration at the south end and oldest area of the site (MW-1) with respect to water quality at the north edge of the site (MW-3).

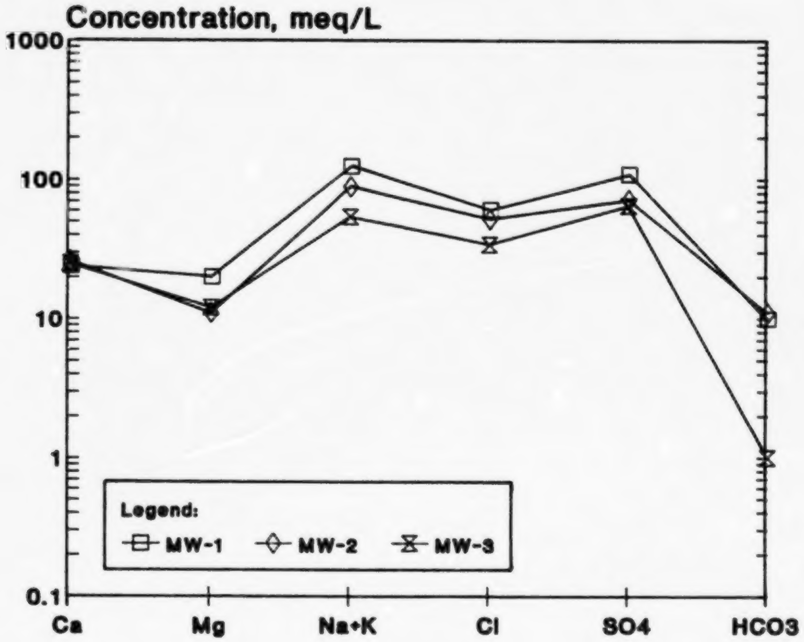


Figure 9-5. Schoeller Diagram: Mean Values From 1987-1989

The G-4 lysimeter water quality data collected in 1968 have been summarized in a Schoeller diagram (Figure 9-6) along with the mean values of samples collected at MW-2 (located in the G-section of the OLF site).

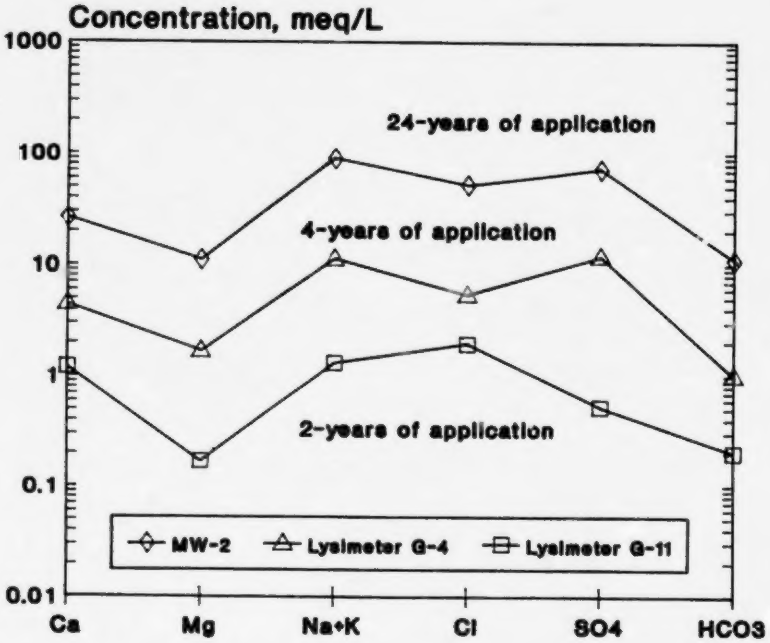


Figure 9-6. Schoeller Diagram: Long-term Groundwater Quality

The data in Figure 9-6 indicate a trend of increased ionic concentrations at the OLF site over time, specifically, below the G-4 line. The similarity in ionic ratios (composition) suggest an apparent "match" between the more recent groundwater data collected at MW-2 and the older data from the lysimeter at G-4. These results strongly suggest that over a long period of time (i.e., ≤ 24-years), groundwater may have evolved from a relatively dilute to a more concentrated solution.

Geochemical Evolution of Groundwater

The Bonham Formation is composed of clays and marls of marine origin (Chapter 8). The porosity of clays and marls is often very high, sometimes exceeding 50%, with very fine and numerous pores that provide an enormous area of contact between water and rock. Although many clays are commonly rated as impermeable and water movement through them is very slow, their colloidal nature and the fineness of their pores permit clays and marls to retain large quantities of water as well as salts (chlorides and sulfates) by adsorption, either by the sediments during deposition or from connate sea water (i.e., water trapped in the clays at the time they were deposited).

A high NaCl content in groundwater does not necessarily indicate connate water, but may simply mean that there has been salt concentration in stagnant groundwater by (a) dissolution of soil minerals or, (b) evapotranspiration of salts in the applied wastewater. The evapotranspirative concentration of salts in the applied wastewater cannot produce a groundwater with the quality found at the OLF site. Based on the evapotranspiration rates measured at the site(10), the concentration of Cl^- in the applied wastewater (Table 5-2), and a mean application rate of 16,000 m^3/d, evapotranspiration alone could only result in a groundwater Cl^- concentration in the range of 100 to 200 mg/L. The lysimeter data (Table 9-6) collected from the root zone support these calculations.

The evapotranspirative concentration trend is demonstrated by the data in Table 7-13, The data indicate that leaching of Cl^-, most likely due to limited infiltration of precipitation, has removed Cl^- from the upper soil surface in the control area. However, significantly higher concentrations (between 115 and 590 mg/L) of Cl^- were measured in surface soil (0 to 30-cm depth) at the the wastewater application areas with respect to the control area (65 mg/L). This pattern may be explained by the evapotranspiration of salts in the applied wastewater by the very dense vegetation at the application areas.

Because evapotranspiration at the OLF site is insufficient to produce the measured concentations of salts in the groundwater, it appears that the dissolution of naturally present minerals may be the primary source of groundwater salinity. Therefore, this subsection includes a discussion of mineral dissolution as the most probable mechanism for the geochemical evolution of groundwater below the OLF site.

The development of groundwater quality below the Campbell Soup OLF can be conceptualized as follows. After the start of OLF wastewater application, some fraction of the treated effluent would infiltrate the soil. Due to the negligible hydraulic gradient of the water table (0.0043 m/m) and the relatively low hydraulic conductivity of the Bonham Formation clays ($k = 10^{-6}$ cm/s), the infiltrated water would tend to remain in the pore spaces or move laterally very slowly. The most rapid transmission of water would occur through the highly permeable weathered contact zone, soil macropores, and fractures in the clays of the Bonham Formation.

Based on the application of Darcy's law to the water level data (mean hydraulic gradient of 0.0043 m/m), an assumed effective porosity of 0.35, and slug test data (k between 10^{-3} and 10^{-4} cm/s) presented previously, the groundwater would travel horizontally at a velocity in the range of 0.4 to 4-m/yr. Thus, groundwater which had infiltrated through the soil to the weathered contact zone at the start of wastewater applications in 1964 would have moved laterally a maximum distance of less than 100-m in the 24-years that the site has been in operation. Because of these low groundwater velocities in the aquifer at the OLF site, stationary groundwater would tend to continually dissolve and eventually approach equilibrium with naturally present minerals in the surrounding clays. Therefore, with time the ionic concentration of the groundwater would tend to increase. This trend is suggested by the data presented in Figure 9-6. A more rigorous description of the theoretical basis for this proposed mechanism for the geochemical development of brackish groundwater below the OLF system is presented in the the following discussion.

From a geochemical viewpoint, the development of a slightly saline aquifer at the Paris, Texas site can be explained in terms of mineral availability and mineral solubility. There are several soluble sedimentary minerals that release SO_4^{2-} or Cl^- upon dissolution. Gypsum ($CaSO_4 \cdot 2H_2O$) and anhydrite ($CaSO_4$) are the most common of the sulfate bearing minerals.

Highly soluble chloride minerals such as halite (NaCl) and sylvite (KCl) often occur as salt strata originally deposited during evaporation of closed or restricted marine basins several million years ago. Chloride minerals of sedimentary origin dissolve rapidly in water. Laboratory analyses of soil samples collected at the OLF site as well as the geologic history of the region confirm the deposition and current presence of these minerals in the Bonham Formation.

The relatively high ionic concentrations of sodium, chloride, and sulfate in the groundwater at the OLF site can be described as the end result of a number of interrelated geochemical processes. The four principal geochemical processes are represented by the reactions:

$$CaSO_4 \cdot 2H_2O \rightarrow Ca^{2+} + SO_4^{2-} + 2H_2O \tag{9-3}$$

$$Ca^{2+} + 2Na(adsorbed) \Leftrightarrow 2Na^{+} + Ca(adsorbed) \tag{9-4}$$

$$CaCO_3 + H_2CO_3 \rightarrow 2Ca^{2+} + 2HCO_3^{-} \tag{9-5}$$

$$NaCl \rightarrow Na^{+} + Cl^{-} \tag{9-6}$$

where (*adsorbed*) denotes cations adsorbed on clays. The dissolution and precipitation of gypsum ($CaSO_4 \cdot 2H_2O$) represents significant mechanisms for the addition or removal of salinity from the groundwater at the OLF site. Because gypsum deposits typically contain magnesium, the dissolution of gypsum will also result in an increase in the concentration of magnesium in groundwater. Although the dissolution and precipitation of gypsum involve equimolar concentrations of $CaSO_4 \cdot 2H_2O$ (equation 9-3), calcium concentrations in groundwater are usually less than that of sulfate since calcium may replace exchangeable sodium (equation 9-4) or magnesium on the soil or precipitate as $CaCO_3$ (equation 9-5).(34)

The exchange of calcium for sodium on the clays of the Bonham Formation is represented by equation 9-4. The equilibrium for equation 9-3 is far to the right as long as there is appreciable Na^+ on the exchange sites of the clays. Equation 9-3 will proceed to the right as long as the activity product $[Ca^{2+}][SO_4^{2-}]$ is less than the equilibrium constant for gypsum and as long as gypsum is available for dissolution.(6) The removal of Ca^{2+} from solution by exchange with Na^+ causes the groundwater to become or remain undersaturated with respect to gypsum, thereby enabling gypsum dissolution to continue. Thus, the influx of Ca^{2+} derived from the dissolution of gypsum causes increased Na^+ (and Mg^{2+}) concentrations as the cation exchange reactions adjust to maintain equilibrium. The increased Ca^{2+} concentrations also cause the solubility product of calcite to be exceeded and drive equation 9-5 to the left resulting in the precipitation of calcite. The precipitation of calcite would cause a loss of Ca^{2+} and HCO_3^-, and dissolution of more gypsum. Because the HCO_3^- content in groundwater is primarily derived from soil zone CO_2 and from dissolution of calcite and dolomite, the removal of HCO_3^- due to precipitation of calcite would result in a decrease in groundwater buffering capacity and with this decrease, the increased probability of a trend towards decreasing pH. Given enough time, a state of equilibrium may be reached and groundwater HCO_3^- concentrations would eventually rise as HCO_3^- equilibrated with atmospheric CO_2.

Because the clay deposits of the Bonham Formation have had the opportunity to contain marine water, the presence of chloride in the soil at the OLF site in not unexpected. The repeated transgression of ancient seas (Chapter 8) in north-central Texas provided conditions suitable for the deposition of sediments which could entrap saline water (connate water). Flushing of the Bonham Formation by rainfall is negligible due to the low permeability of the clays and the lack of significant quantities of precipitation. The lack of deep percolation would permit chloride in the soil to remain near the land surface after millions of years of potential flushing. With the initiation of land treatment operations, excess quantities of water beyond the crop requirements have been supplied. The excess water can percolate down through the clay and dissolve the ancient chloride deposits.

Only a very small amount of chloride need be present in the clay for relatively high concentrations to develop in the groundwater. Because chloride generally does participate in chemical or biological reactions within the groundwater systems, it is often considered to be a conservative substance. As such, once dissolution of NaCl occurs, Cl^- will remain in solution, unaffected by precipitation, ion exchange, or biochemical degradation. At equilibrium, the soil pore water concentration of Cl- is the same concentration (per unit volume) as the soil concentration of Cl^-. Analysis of unleached soil samples collected from the control area (Table 7-13) indicated that the soil pore water Cl^- concentration within the unsaturated zone of the Bonham clay was roughly equivalent to the mean Cl^- content of the groundwater. Thus, it appears probable that the natural mineral content of the Bonham Formation contains enough Cl^- to produce the Cl^- concentraion found in the groundwater at the site.

The groundwater monitoring data collected at MW-1, MW-2, and MW-3 support the preceeding discussion of the development aqueous geochemistry at below the Campbell Soup OLF site. Water samples collected from the older areas (i.e., areas with over 24-years of wastewater application) have had a correspondingly long period of time to progress towards aqueous geochemical equilibrium; therefore, the relatively high concentrations of Na^+, SO_4^{2-}, Cl^-, and HCO_3^- measured in groundwater samples collected at MW-1 and MW-2 can be expected. However, in the newer areas near MW-3, groundwater has had significantly less time (<10-years) to equilibrate with clay minerals and atmospheric CO_2. Thus, the ionic concentrations of the major cations at MW-3 are expected to be slightly less than those at MW-1 and MW-2. The field data (Table 9-1, 9-2, and 9-3) support this prediction, and the low concentration of HCO_3^- in the water at MW-3 provide further evidence of the relatively recent recharge of treated wasetwater and the development of the groundwater chemistry at the site.

Summary of the Conceptual Model for Groundwater Recharge

Based on the findings of the hydrogeologic study, a conceptual model of the development and recharge of an semi-confined aquifer below the site was developed (Figure 9-7). In this study, the aquifer system below the OLF site was conceptualized as having one primary flow system, through fractures and along the erosional contact with slow recharge through the pore spaces of the Bonham Formation clays. These flow systems exist in the heavy, inorganic clay and marl of the Bonham Formation (a unit of the Austin Chalk Group), and are confined from below by the relatively impermeable Maribel Shale, the upper part of the Eagle Ford Group.

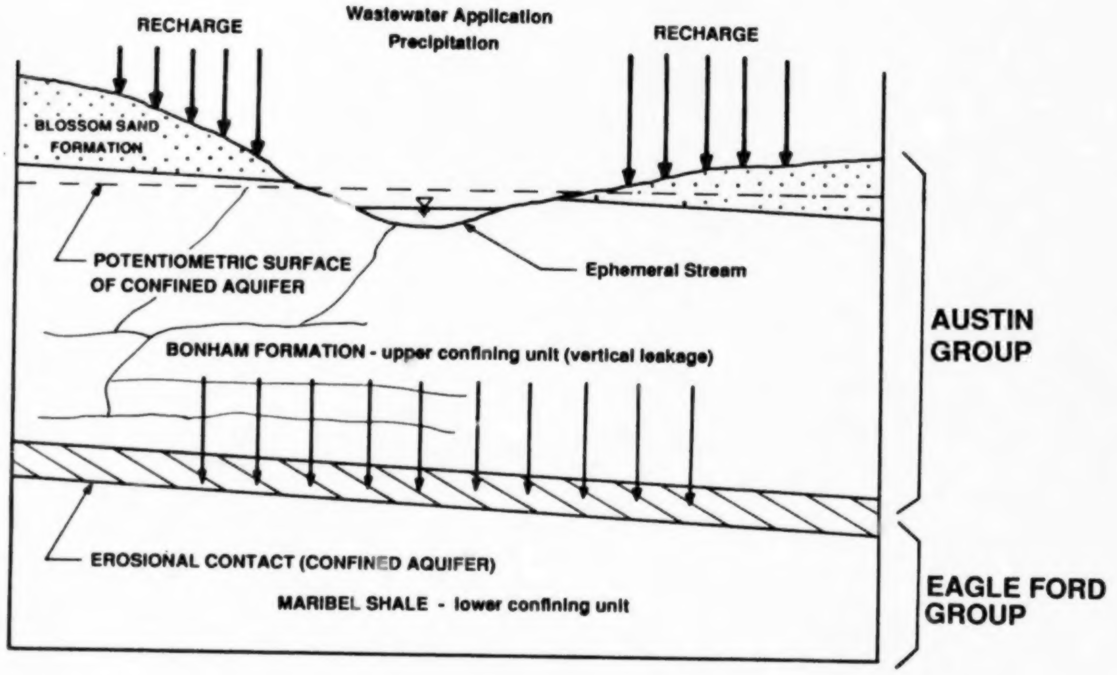

RECHARGE
Wastewater Application
Precipitation
RECHARGE
BLOSSOM SAND FORMATION
POTENTIOMETRIC SURFACE OF CONFINED AQUIFER
Ephemeral Stream
AUSTIN GROUP
BONHAM FORMATION - upper confining unit (vertical leakage)
EROSIONAL CONTACT (CONFINED AQUIFER)
MARIBEL SHALE - lower confining unit
EAGLE FORD GROUP

Figure 9-7. Conceptual Model of the Groundwater Recharge

As shown in Figure 9-7, the aquifer is recharged slowly by water (mostly treated wastewater with some precipitation) moving vertically downward through pore spaces and fractures within the Bonham Formation. Within the aquifer, the principal zone of high hydraulic conductivity is located at the erosional contact between the Austin and Eagle Ford Groups. Preferential flow within fractures and macropores may represent a secondary pathway for the transmission of groundwater.

The evapotranspirative concentration of salts in the deep percolate may result in above "background" accumulations of salts at the soil surface, even though leaching would continue to occur. The combined effects of mineral dissolution and evapotranspiration would result in increases in the salinity of the porewater. As a result of the lack deep percolation (due to low precipitation and the low hydraulic conductivity of the clays), readily soluble minerals had remained in the soil until the initiation of OLF operations. Water moving down through the soil will leach these naturally present, highly soluble minerals from the Bonham Formation.

In addition, the application and subsequent infiltration of large volumes of wastewater may result in the development of new solution channels as well as the enlargement of existing channels. These channels can act as conduits for the seepage of large volumes of water through the subsurface.

Due to the negligible dip of the lower confining unit (Eagle Ford Group) and the relatively flat regional surface topography in the area, a negligible hydraulic gradient would result. Because of the small hydraulic gradient and the low hydraulic conductivity of the Bonham Formation within the pore spaces and along fractures, infiltrated water would tend to remain relatively stationary once a unit area reaches saturation. Thus, the stationary water would dissolve and remove minerals from the surrounding clay until the water moved on or the soil-water system reached a point of stability (chemical equilibrium). The rate of groundwater movement in the Bonham Formation over the entire area of the OLF site is expected to be uniformly low. Therefore, groundwater quality within the Bonham Formation at the OLF site would tend to be similar. However, groundwater quality at different locations are not expected to be of identical quality because of the extremely slow rate of flushing. Localized concentrations of specific minerals, such as gypsum or sodium chloride, may result in nonuniform concentrations of these constituents in the groundwater within the Bonham Formation. This mechanism of groundwater recharge and mineral dissolution is hypothesized to have resulted in the development of moderately saline aquifer below the OLF site.

Assuming water table conditions, Smith Creek would penetrate the surface of the aquifer at topographically low regions of the OLF site. These low areas may also correspond to groundwater discharge areas. However, it is not expected that significant discharge to the Smith Creek would occur due to the low hydraulic conductivity of the saturated Bonham Formation clays. These predictions are supported by the fact that long-term water quality monitoring data collected at the Smith Creek monitoring station (Chapter 6) did not indicate elevated concentrations of sodium, calcium, sulfate, and chloride in the Creek. Thus, because groundwater discharge as baseflow is predicted to be negligible and there are no springs within the vicinity of the OLF site nor are there pumped wells screened within the aquifer, the major mechanism for aquifer discharge is unknown. However, due to the relatively low hydraulic conductivity of the Bonham Formation, groundwater would tend to travel at such a slow rate that the discharge would be difficult to identify either visually or by influences on the chemical quality of receiving water bodies. In addition to surface discharge as baseflow and springs, the groundwater may discharge by seepage into Lake Crook, to the north, or by discharge into other formations in the subsurface along faulting planes and fracures.

SUMMARY

Twenty six groundwater samples collected from three monitoring wells (MW-1, MW-2 and MW-3) between 1987 and 1989 were analyzed for the constituents listed in Appendix B. In general, the water was moderately saline and no purgable and extractable organics were detected. Statistical analysis (ANOVA at $p \leq 0.05$) of the data indicated that, for the major ions present, groundwater quality at each well remained uniform over the 3-year period of sampling. However, statistically significant differences in groundwater quality between the well locations were found. The consistency of groundwater quality at each location permitted the use of mean values for comparisons between wells and for the graphical presentation of data.

State records indicated that no wells in Lamar County were screened within the Austin Chalk-Eagle Ford Group. In addition, no springs which issued from the Bonham Formation were identified within Lamar County or any adjacent counties.

Several graphical techniques were used to visually examine the similarity in water quality among the three monitoring well locations and the lysimeter data. Schoeller plots of major-ion concentrations indicated that water collected from all three monitoring wells exhibited similar ionic composition ratios. These waters appeared to have undergone similar paths of geochemical evolution with a trend towards increasing ionic concentration from the north edge of the OLF site to the south edge. This trend is in agreement with the relative length of time that the north (<10-years) and south (>20-years) areas have been in operation. Ionic ratios of major ions in groundwater data collected from lysimeters at the site in 1968 exhibited excellent agreement with comparable ionic ratios for more recent samples from MW-2 (1987-1988). The pattern of increasing ionic compostion with relatively small changes in ionic ratios strongly suggested a trend towards the concentration of dissolved minerals in the relatively slow moving groundwater.

The geochemical data indicated that sulfate-chloride facies were dominant for groundwater at all three monitoring wells (MW-1, MW-2 and MW-3) and for the lysimeter data collected in 1968. Thus, the similarity of water quality among well locations as well as the similarity in the pattern of geochemical evolution over the past 24-years was strongly suggested.

The infiltration of treated wastewater, development of water-table aquifer, the dissolution of soluble minerals from the soil and resultant effects on the quality of groundwater below the site were strongly suggested by the groundwater geochemical data, soil data, the expected rate of infiltration, field hydraulic conductivity and water level measurements, and the magnitude of the volume of wastewater applied.

CHAPTER 10

GLOSSARY OF TERMS

Absorption

Taking up, incorporation, or assimilation, as of liquids in solids or of gases in liquids.

Adsorption

Adhesion of gas molecules, or of ions or molecules in solution to the surface of solid bodies with which they are in contact.

Alluvium

Detrital deposits made by streams on river beds and flood plains; esp. a deposit of silt or silty clay laid down during a time of flood. The term applies to deposits of recent origin. It does not include subaqueous sediments of seas and lakes.

Aquiclude

A body of rock that will absorb water slowly but will not transmit it fast enough to supply a well or spring.

Aquifer

A body of rock that is sufficiently permeable to conduct groundwater and to yield economically significant quantities of water to wells and springs.

Aquitard

A confining bed that retards but does not prevent the flow of water to or from an adjacent aquifer; a leaky confining bed. It does not readily yield water to wells or springs, but may serve as a storage unit for groundwater.

Aragonite

An orthorhombic mineral, $CaCO_3$, trimorphous with calcite and vaterite. It occurs in beds of gypsum and iron ore, and in shallow marine banks, an in pearls and some shells.

Barite

An orthorhombic mineral, $BaSO_4$, with a specific gravity of 4.5 It is the principal ore of barium.

Baseflow

That part of stream discharge from groundwater seeping into the stream.

Calcite

A common rock forming mineral, $CaCO_3$. Commonly white or gray, it has perfect rhombohedral cleavage and reacts readily with cold dilute hydrochloric acid. Calcite is the chief constituent of limestone and most marble.

Celestite

An orthorhombic mineral, $SrSO_4$. The principal ore of strontium.

Clastic

Pertaining to a rock or sediment composed principally of fragments derived from pre-existing rocks or minerals and transported some distance from their place of origin; also said of the texture of such a rock.

Common-ion effect

The decrease in solubility of a salt dissolved in water already containing some of the ions of the salt.

Confined aquifer

A aquifer bounded above and below by impermeable beds, or by beds of distinctly lower permeablity than that of the aquifer itself; an aquifer containing confined groundwater.

Confined groundwater

Groundwater that is under suffcient pressure to rise above the level at which it is encountered in a well; it may or may not flow to or above the ground surface. Its upper surface is the bottom of an impermeable bed.

Connate water

Water entrapped in the interstices of a sedimentary rock at the time the rock was deposited.

Darcy's law

An equation that can be used to calculate the quantity and velocity of water moving through a porous media.

Deltaic

Pertaining to or characterized by a delta; e.g. "deltaic sedimentation.

Diaspore

A gray or yellowish orthorhombic mineral, AlO(OH), dimorphous with boehmite, It is found in bauxite deposites.

Dip

The angle that a stratum or any planar feature makes with the horizontal, measured perpendicular to the strike and in the vertical plane.

Discharge area

An area in which there are upward components of hydraulic head in an aquifer. Groundwater is flowing toward the surface in a discharge area and may escape as a spring, seep, or baseflow or by evaporation and transpiration

Dolomite

A common rock forming mineral, $CaMg(CO_3)_2$. Part of the magnesium may be replaced ny ferrous iron. Dolomite is light colored and has perfect rhombohedral cleavage. Most dolomite is associated and often interbedded with limestone.

Effective porosity

The percent of the total volume of a given mass of soil or rock that consists of interconnecting voids.

Electromagnetic survey

A geophysical method employing the generation of electromagnetic waves at the earth's surface; when the waves impinge on a conducting formation (such as saline groundwater or an ore body) at depth they induce currents that are the source of new waves radiated from the conductors and detected by instruments at the surface.

Ephemeral stream

A stream or portion of a stream which flows briefly in direct response to precipitation in the immediate vicinity, and whose channel is at all times above the water table.

Evaporite

One of the sediments which are deposited from aqueous solution as a result of extensive or total evaporation. Examples include anhydrite, rock salt, various nitrates and borates.

Evapotranspiration

That portion of the precipitation returned to the air through evaporation and plant transpiration.

Fissile

Capable of splitting easily along closely spaced parallel planes, e.g. bedding in shale.

Glauconite

A green mineral closely related to the micas and essentially a hydrous potassium iron silicate. It is common in sedimentary rocks from Cambrian to the present. It is an indicator of very slow sedimentation.

Gypsum

A widely distributed mineral consisting of hydrous calcium sulfate: $CaSO_4 \cdot 2H_2O$. It is the commonest sulfate mineral, and is frequently associated with halite and anhydrite in evaporites, forming thick, extensive beds.

Hiatus

A break or interruption in the continuity of a stratigraphic record, such as the absence of rocks that would normally be present in a sequence but were never deposited.

Humic acid

Black acidic organic matter extracted from soils, low rank coals, and other decayed plant substances by alkalis. It is insoluble in acids and organic solvents.

Hydraulic conductivity

The rate at which water flows in m/d through a cross section of one square meter under a unit hydraulic gradient.

Ion exchange

A process by which an ion in a mineral lattice is replaced by another ion that was present in solution.

Inorganic

Pertaining or relating to a compound that contains no carbon.

Lamina

The thinnest recognizable layer in a sediment or sedimentary rock, differing from other layers in color, composition, or particle size; and commonly 0.05 to 1.00 mm thick.

Lithology

The description of rocks on the basis of such a characteristics as color, mineralogic composition, and grain size.

Marl

A term loosely applied to a variety of materials, mostly unconsolidated earthy deposits chiefly consisting of an intimate mixture of clay and calcium carbonate, usually including shell fragments and sometimes glauconite. It is formed under marine and especially freshwater conditions.

Megafossils

Fossils that are large enough to be seen with the unaided eye.

Neutron probe log

A radioactivity log that indicates the intensity of radiation produced when rocks or soil in a borehole are bombarded by neutrons. It indicates the presence of fluids.

Perched groundwater

Unconfined groundwater separated from the underlying main body of groundwater by unsaturated rock.

Pore space

The volume between mineral grains in a porous medium.

Potentiometric surface

An imaginary surface representing the total head of groundwater and defined by the level to which water will rise in a well.

Recharge area

An area in which there are downward components of hydraulic head in an aquifer. Infiltration moves downward into the deeper parts of an aquifer in a recharge area.

Saturated zone

The zone in which the voids in the rock or soil are filled with water at a pressure greater than atmospheric. The water table is at the top of the saturated zone in an unconfined aquifer

Secondary porosity

The porosity developed in a rock after its deposition or emplacement, through such processes as solution or fracturing.

Semiconfined aquifer

An aquifer confined by a low-permeability layer that permits water to slowly flow through it. During pumping of the aquifer, recharge to the aquifer can occur across the confining layer. Also known as a leaky confined aquifer.

Series

A chronostratigraphic unit next in rank below system and above stage; the rocks formed during an epoch of geologic time.

Shale

A fine-grained detrital sedimentary rock, formed by the compaction of clay , silt, or mud. It has a finely laminated structure, which gives it fissility along which the rock splits readily, especially on the weathered surfaces.

Solubility product

The equilibrium constant that decribes a solution of a slightly soluble salt in water.

Strike

The direction taken by a structural surface, e.g. a bedding or fault plane as it intersects the horizontal.

Transgression

The spread of the sea over land areas; also, any change that brings offshore, deep-water environments to areas formerly occupied by nearshore, shallow-water condtions.

Unconformable

Said of strata that do not succeed the underlying rocks in immediate order of age or in parallel position.

Unsaturated zone

The zone between the land surface and the water table. It includes the root zone, intermediate zone, and capillary fringe. The pore spaces contain water at less than atmospheric pressure, as well as air and other gases. Also called the zone of aeration and the vadose zone.

Water table

The surface of an unconfined aquifer or confining bed at which the pore water pressure is atmospheric.

Weathering

The destructive processes by which rocks are changed on exposure to atmospheric agents at or near the earth's surface, with little or no transport of the loosened or altered material.

Well, fully penetrating

A well drilled to the bottom of an aquifer, constructed in such a way that it withdraws water from the entire thickness of the aquifer.

Well, partially penetrating

A well constructed in such a way that it draws water directly from a fractional part of the total thickness of the aquifer. The fractional part may be located at the top or the bottom or anywhere in between in the aquifer.

CHAPTER 11

CONVERSION FACTORS

METRIC TO U.S. UNITS

Metric Name	Symbol	Multiplier	Symbol	U.S. Name
centimeter	cm	0.3937	in.	inch
cubic meter	m^3	264.17 35.31	gal ft^3	gallons cubic feet
degrees Celsius	°C	1.8(°C) + 32	°F	degrees Fahrenheit
gram	g	0.0022	lb	pound
hectare	ha	2.4711	ac	acre
kilogram	kg	2.205	lb	pound
kilometer	km	0.6214 3,281 1,094	mi ft yd	mile feet yard
liter	L	0.0353 0.2642	ft^3 gal	cubic foot gallon
metric ton	t	1.10 2205	ton lb	ton (short) pounds
meter	m	3.2808 39.37	ft in	foot inch
kiloNewtons per square meter, kiloPascal	kN/m^2, kPa	1.4498 x 10^{-1}	lb$_f$/in^2, psi	pounds per square inch

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APPENDIX A

WELL CONSTRUCTION DATA

WELL CONSTRUCTION DATA

UNIVERSITY OF TEXAS AT AUSTIN
ENVIRONMENTAL ENGINEERING PROGRAM

Client: U.S.EPA	Site: Paris, Texas
Job No.: 68-01-7266	Well No.: MW-1
Well Use: Monitoring	Page 1 of 1

Drilling Summary

Total Depth: 5.49-m	Drillers: U.S.EPA Ada, OK
Borehole Diameter: 10.2-cm	
Rig Type: CME Rotary Auger	Bit(s): Hollow Stem
Elevation (MSL)	Top of Casing: 152.64-m
Land Surface: 152.26-m	Water Level: 151.73-m

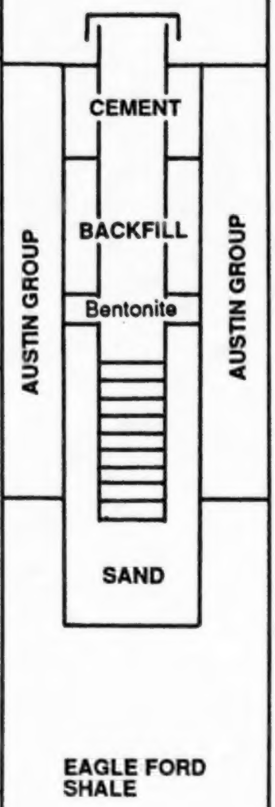
Well Design

Casing Material:Stainless Steel 316	Diameter: 5.1-cm Length: 3.28-m
Screen Material:Stainless Steel 316	Diameter: 5.1-cm Length: 1.52-m
Slot Size: 0.025-cm	Setting: 149.36 to 147.84-m
Seals Material: Bentonite	Setting: 149.67 to 149.97-m
Filter Material: Washed Sand	Setting: 146.77 to 149.67-m
Grout: Portland Cement	Setting: 152.26 to 151.34-m
Surface Casing/Cap:Stainless Steel	Setting: 152.26 to 152.64-m

Well Development:

Method: Vacuum lift system

Remarks: No forced flushing, after development, well produced about 19-L of water per hour.



WELL CONSTRUCTION DATA

UNIVERSITY OF TEXAS AT AUSTIN
ENVIRONMENTAL ENGINEERING PROGRAM

Client: U.S.EPA
Site: Paris, Texas
Job No.: 68-01-7266
Well No.: MW-2
Well Use: Monitoring
Page 1 of 1

Drilling Summary

Total Depth: 7.53-m
Drillers: U.S.EPA Ada, OK
Borehole Diameter: 10.2-cm

Rig Type: CME Rotary Auger
Bit(s): Hollow Stem
Elevation (MSL)
Top of Casing: 153.08-m
Land Surface: 152.65-m
Water Level: 152.06-m

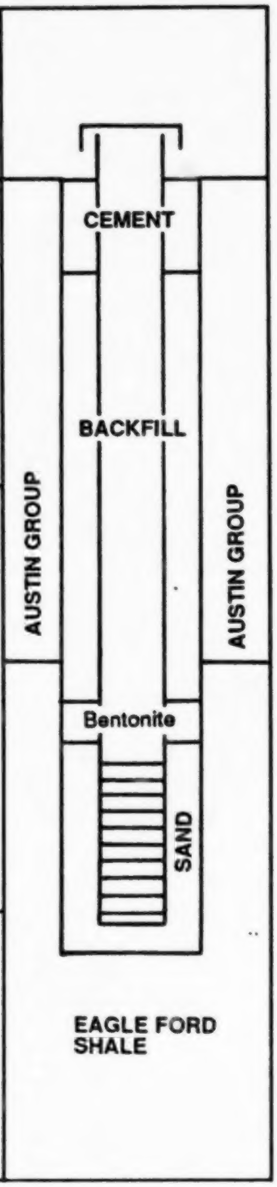
Well Design

Casing Material:Stainless Steel 316
Diameter: 5.1-cm
Length: 6.13-m
Screen Material:Stainless Steel 316
Diameter: 5.1-cm
Length: 1.52-m
Slot Size: 0.025-cm
Setting: 146.95 to 145.45-m
Seals Material: Bentonite
Setting: 147.55 to 147.25-m
Filter Material: Washed Sand
Setting: 147.25 to 145.15-m
Grout: Portland Cement
Setting: 152.65 to 151.75-m
Surface Casing/Cap:Stainless Steel
Setting: 152.65 to 153.08-m

Well Development:

Method: Vacuum lift system

Remarks: No forced flushing, after development, well produced 19-L of water per hour.



WELL CONSTRUCTION DATA

UNIVERSITY OF TEXAS AT AUSTIN
ENVIRONMENTAL ENGINEERING PROGRAM

Client: U.S.EPA	Site: Paris, Texas
Job No.: 68-01-7266	Well No.: MW-3
Well Use: Monitoring	Page 1 of 1

Drilling Summary

Total Depth: 7.82-m	Drillers: U.S.EPA Ada, OK
Borehole Diameter: 10.2-cm	
Rig Type: CME Rotary Auger	Bit(s): Hollow Stem
Elevation (MSL) Land Surface: 142.51-m	Top of Casing: 142.71-m Water Level: 142.68-m

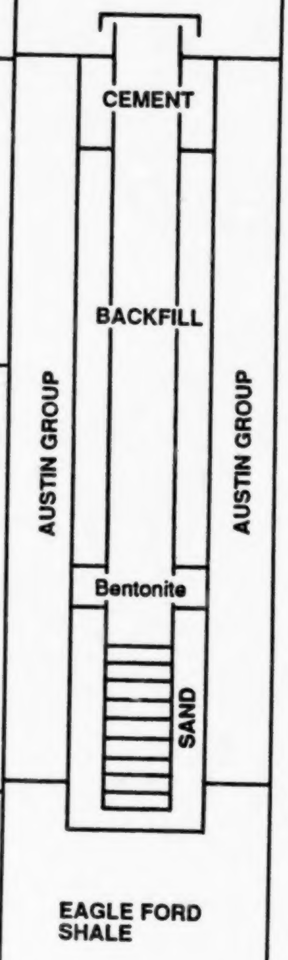
Well Design

Casing Material:Stainless Steel 316	Diameter: 5.1-cm Length: 5.84-m
Screen Material:Stainless Steel 316	Diameter: 5.1-cm Length: 1.52-m
Slot Size: 0.025-cm	Setting: 136.71 to 135.20-m
Seals Material: Bentonite	Setting: 137.48 to 137.18-m
Filter Material: Washed Sand	Setting: 137.18 to 134.89-m
Grout: Portland Cement	Setting: 142.51 to 141.61-m
Surface Casing/Cap:Stainless Steel	Setting: 142.51 to 142.71-m

Well Development:

Method: Vacuum lift system

Remarks: No forced flushing, after development, well produced 19-L of water in 4 minutes.



APPENDIX B

GROUNDWATER CONSTIUENTS ANALYZED

Inorganic Cations

Aluminum
Arsenic
Barium
Beryllium
Boron
Cadmium
Calcium
Chromium
Cobalt
Copper
Iron
Lead
Lithium
Magnesium
Manganese
Nickel
Potassium
Silver
Selenium
Sodium
Strontium
Thallium
Titanium
Vanadium
Zinc

Extractable Organic Compounds

n-Nitrosodimethylamine
Phenanthrene1,2-Diphenylhydrazine/azobenzene
Benzidine
1,3-Dichlorobenzene
1,4-Dichlorobenzene
1,2-dichlorobenzene
Hexachloroethene
bis(2-Chloroisopropyl)ether
bis(2-Chloroethyl)ether
n-Nitrosodi-n-propylamine
Nitrobenzene
Hexachlorobutadiene
1,2,4-Trichlorobenzene
Naphthalene
bis(2-Chloroethoxy)methane
Isophorone
Hexachlorocyclopentadiene (HCCP)
2-Chloronaphthalene
Acenaphthlene
Acenaphthene
Dimethyl phthalate
2,4-Dinitrotolulene
2,6-Dinitrotolulene
4-Chlorophenyl phenyl ether
Fluorene
Diethyl phthlate
n-Nitrosodiphenylamine
Hexachlorobenzene (HCB)

4-Bromophenyl phenyl ether
Anthracene
Di-n-butylphthlate
Fluoranthene
Pyrene
Benzyl butyl phthlate
Bis(2-ethylhexyl)phthlate
Benzo(a)anthracene
Chrysene
3,3'-Dichlobenzidine
Di-n-octylphthlate
Benzo(b and/or k)fluoranthene
Benzo-a-pyreneIndeno (1,2,3-cd)
pyrene
Dibenzo(a,h)anthracene
Benzo(ghi)perylene
2-Chlorophenol
2-Nitrophenol
Phenol2,4-Dimethylphenol
2,4-Dichlorophenol
2,4,6-Trichlorophenol
4-Chloro-3-methylphenol
2,4-Dinitrophenol
2-Methyl-4,6-dinitrophenol
Pentachlorophenol
4-Nitrophenol

Miscellaneous Extractable Organics

Hexanoic acid
Heptanoic acid
Octanoic acid
Decanal
Nonanoic acid
Decanoic acid
Dodecanoic acid
Tetradecancic acid
Tridecanoic acid
Pentadecanoic acid
Hexadecanoic acid
Octadecanoic acid1-
Hexadecanol, acetate
n,n-Diethyl-3-methyl-benzamide
C3-Benzene
Heptanal
9-Hexadecenoic acid
Dichlorbenzene (3 isomers)
Benzothiazole
[1,1'-Biphenyl]-2-ol
Isobutyl stearate
Isobutyl palmitate
Hexadecanol
3-(1-Methyl-2-pyrolidinyl)-pyridine

Purgeable Organic Compounds

Methylene chloride
1,1-Dichloroethene
1,1-Dichloroethane
trans-1,2-Dichloroethene
Chloroform
1,2-Dichloroethane
1,1,1-Trichloroethane
Carbon tetrachloride
Bromodichlormethane
Trichloroethene
Benzene
Bibromochloromethane
1,1,2-Trichloroethane
Bromoform

Tetrachloroethane

Toluene

Chlorobenzene

Ethyl benzene

m-Xylene

o & p-Xylene (mixed)

Inorganic Anions

Chloride

Ortho-phosphate phosphorous

Sulfate

NO_3-NO_2-N

Additional Analyses

Conductivity

Alkalinity

Ammonia

pH

TOC

APPENDIX C

SLUG TEST RESULTS

Distance to top of casing, ft-in.	Elapsed time, sec
SLUG TEST No. 1	MW-3 7-18-89
2' 9"	0.0
2' 3.5"	20
2' 2"	30
1' 9"	40
1' 7"	50
1' 5"	60
1' 4"	70
1' 3"	80
1' 2"	90
1' 1"	100
SLUG TEST No. 2	MW-3 7-18-89
4' 1"	0.0
3' 6"	10
2' 8"	30
2' 4"	40
2' 2.5"	50
1' 9"	60
1' 7"	70
1' 6"	80

Distance to top of casing, ft-in.	Elapsed time, sec
SLUG TEST No. 3	MW-3 7-18-89
3' 4"	5
3' 7.5"	15
3' 2.5"	25
2' 8.5"	33
2' 6"	42
2' 4.5"	50
2' 1.5"	58
2' 0.5"	65
1' 8"	80
1' 6"	90
1' 5"	100
1' 4"	110
1' 3"	120
SLUG TEST No. 4	MW-3 7-18-89
4' 6"	0
4' 0"	11
3' 6"	18
3' 3"	24
3' 0"	34
2' 6.5"	40
2' 5"	50
2' 3"	57
2' 1"	60
2' 0"	70
1' 8"	80
1' 6"	90
1' 5"	100
1' 4"	110

Distance to top of casing, ft-in.	Elapsed time, sec
SLUG TEST No. 5	MW-2 7-18-89
9' 1"	0
8' 7"	50
8' 4"	90
8' 2"	120
8' 0"	150
7' 8"	180
7' 6"	210
7' 4.5"	240
7' 4"	270
SLUG TEST No. 6	MW-2 7-18-89
9' 6"	0
9' 4"	20
9' 2"	40
9' 0"	70
8' 8"	115
8' 6"	150
8' 4"	195
8' 2"	250
8' 0"	305
7' 8"	375